Ecological Restoration and Management of the Linwood Paddocks

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CONTENTS

ABSTRACT

This study investigates salt marsh restoration within the Canterbury region as a way to remedy the current loss of these vital systems. Previous attempts have concentrated largely on aesthetic appeal and lack detailed scientific basis or understanding. The proposed Linwood Paddock restoration site (historically a brackish wetland) provides an excellent opportunity for scientifically-based salt marsh restoration. Restoration should reestablish functions and attributes symptomatic of a 'healthy' wetland system, from the presently imposed pastoral system.

The historic and current salt marsh composition and relation to environmental conditions are described for the Canterbury region and the long-term impacts of implementation and management decisions in restoration are evaluated. The likely success of plant growth on heavy metal and nutrient-enriched Linwood soils and the potential for phytoremediation to improve wastewater quality are also determined.

Quantitative survey's of the fragmented Avon, Heathcote, Brooklands and Saltwater Creek salt marshes, provided a complete revegetation template, which details the optimal range for each native species with respect to elevation and salinity. The marked zonation appeared to be the result of both competitive displacement and interspecific variation in physiological tolerance. Lower marsh elevations were dominated by *Leptocarpus similis* and upper marsh elevations by *Sarcocornia quinqueflora*. All other species' requirements overlap, but they would be expected to exist in a mid tidal mosaic "patchwork" fashion in revegetation, since the clonal nature of these species means that patches will be monospecific, rather than mixed-species patches.

A mesocosm experiment was performed in an area of the Heathcote marsh by planting *Leptocarpus similis*, *Juncus maritimus* and *Schoenoplectus pungens* into transferred Linwood Paddock soil. Plant growth was viable in Linwood Paddock topsoil and plants sourced from natural stocks had significantly higher survival and biomass than nursery stock ($p < 0.05$). In addition, there was a significant difference ($p < 0.05$) in survival and biomass between the two species remaining at the end of

the experiment, with *Leptocarpus similis* having increased survival and greater biomass than *Juncus maritimus*.

Assessments of previous local revegetation attempts at the margins of Oxidation Pond Nos. 5 and 6, the Charlesworth Street Reserve and the Devil's Elbow bankworks showed that salt marsh herbs (*Sarcocornia quinqueflora* in particular) were more likely to self-colonise a restoration site than rushes or sedges. The most obvious impediments to revegetation success were the use of a too coarse substrate and insufficient tidal flushing through lack of maintenance. It was concluded that adequate monitoring and record-keeping are necessary to determine strategies for wetland implementation and to understand the long-term impacts of wetland management decisions from such attempts.

The scientific research was integrated in the design of an ecological salt marsh restoration and management plan for the Linwood Paddocks, which is not only representative of Canterbury salt marshes, but also self-sustaining and accessible to the general public. Creation of a freshwater wetland within the existing Bromley Oxidation Ponds is presented as a unique opportunity to remediate wastewater and to supply high quality freshwater to the salt marsh restoration, thereby enhancing estuary wildlife and water quality values simultaneously. The freshwater species recommended for phytoremediation are *Typha* and *Scirpus*.

This design is scientifically based and its principles could be used (with the application of local data) in further restoration projects within New Zealand or elsewhere in the temperate world. As salt marshes decline, this approach is timely to ensure that restoration is successful and not a technique that is relearned each time.

1. GENERAL INTRODUCTION

Salt marshes occur throughout the temperate world and comprise areas of land bordering the sea, more or less covered with vegetation and subject to periodic inundation by the tide (Chapman, 1974).

Although salt marshes are important in terms of terrestrial and aquatic production, shoreline stabilisation and climate regulation, their association with human civilization has been one of fear, continued degradation, pollution and reclamation.

There has been little research on salt marsh distribution and its relation to environmental parameters in New Zealand. Even this little information has rarely been applied to salt marsh protection or attempts to regain those areas that have been lost. Restoration requires a scientific basis detailing wetland vegetation, hydrology, soils and wildlife. It also needs to recognise engineering and economic constraints. The retention and restoration of salt marshes is particularly important in New Zealand (a small country with a large maritime influence) for providing protection against sea level rise, water purification, provision of habitat for native and migratory birds, conservation of native plant species, and as feeding areas for recreational and commercial fishing stocks.

This thesis attempts to determine the historic and current distribution of salt marsh vegetation in Canterbury with respect to environmental parameters. The implications for salt marsh restoration on the proposed Linwood Paddock restoration site will be examined in an attempt to formulate a restoration plan specifically for this area. The council-owned Linwood Paddocks are located adjacent to the Avon-Heathcote Estuary and the Bromley Oxidation Ponds. The low-lying paddocks were formed after drainage and leveling in the 1900s, and their subsequent use has been for pastoral farming and application of biosolids from the Bromley sewage treatment works. Concerns about such practices mean that their continuation is under review. Restoration of the paddocks should allow the system to revert from the current pastoral use to a functioning salt marsh ecosystem - analogous to that found "naturally" in the Canterbury region.

1.1 Wetland Perception and Loss

Despite their numerous values and functions beneficial to man, wetlands are regarded and treated by many as wastelands. Few are aware that human society developed on the water's edge, where freshwater and the richest farmland was found on valley bottoms and floodplains, and where the most accessible fishing and shell-fishing was in estuaries and coastal waters fed by tidal marsh and mud flat (Williams, 1994). Indeed, wetlands played a key role in sustaining early prehistoric cultures such as the Mesolithic occupants of postglacial lake margins and coasts in Europe (Maltby, 1988) and the major civilizations of the floodplain environments of the Nile, Tigris and Euphrates. Today there is still a strong dependency on wetland resources by communities in the developing Third World. Unfortunately, the history of wetlands in the developed world has been one of progressive detachment from direct human civilization and conversion to non-wetland uses (Maltby, 1988). To our detriment, human adaptation to wetlands has been superseded by exploitation via reclamation. Such activity is referred to as "reclaiming" rather than "claiming", implying that all wetlands should be dry lands and that mankind has been merely correcting nature's mistake (Williams, 1994).

Initially we made use of wetlands by adaptation. We no-longer adapt, but exploit, a new idea derived from the Industrial Revolution's utilitarian views of nature that demand resource exploitation for economic development (Williams, 1994). Certainly, Sauer (1938) regarded traditional wetland utilization as a 'passing frontier of nature replaced by a permanently and sufficiently expanding frontier of technology'. He believes this attitude has the recklessness of an optimism that has become habitual. We forget that by conquering nature and by "winning" the battle against it, we surely find ourselves on the losing side. Reclamation of wetlands is the lot of the losing team.

Regrettably, the world has a belated appreciation of the 2 % of the total Earth's surface that comprise wetlands (Williams, 1994). The general lack of government interest, financial support and, until recently, scientific identity or study (Maltby, 1988) has lead to numerous misconceptions surrounding wetlands.

Negative connotations ensure that wetland destruction and alteration has prevailed over an alternative view of maintenance and enhancement. For example, prevalence of the 'wasteland' concept and the belief that drainage and conversion is a 'public-spirited endeavour' (Baldock, 1984), has led to agricultural conversion accounting for 87 % of wetland losses in the United States (Maltby, 1988). Conversion of wetland to dry land indubitably achieves a higher economic value for a very specific purpose, such as farming or urban development and for specific interest groups i.e. farmers and land developers (Williams, 1994). These benefits are tangible and usually quickly realised, but they involve huge costs which are far less tangible because they are environmental and cumulative, and can take a long time to be translated into economic costs (e.g. loss of fish spawning areas will eventually decrease local fisheries, though it may take several seasons before a loss is noted). These costs are seldom borne by the beneficiaries of reclamation alone, but by the taxpayer or society at large and the global ecosystem (Williams, 1994).

The association with disease especially malaria and schistosomiasis, and also the physical danger in traversing such areas have been further motives for wetland drainage. England was malarial until the 1400s when climate change and progressive wetland drainage lead to the loss of many areas including the English fens (Darby, 1983). The 6,000 ha Hula Papyrus swamp in northern Israel in the 1950s was also drained with these fears in mind. Even in New Zealand today, wetland areas are despised and feared following the finding of the Ross River fever virus carried by mosquitos, which breed in some wetlands.

Very long-term positive functional values are rarely considered and the complexity, integrity, and uniqueness of natural wetlands are undervalued (Zedler and Weller, 1990). The protection and filtering of water (our most vital natural resource), the critical role in maintaining biodiversity, the capacity as a carbon sink that buffers global warming, or the importance of salt marshes in supporting valuable coastal fisheries (as well as protection against coastal flooding) are all too often forgotten.

Surely, the precautionary principle of sustainable development applied to the slow recovery and practical unrecreatability of many wetlands, coupled with the unknown quantum of their contribution to biodiversity and the local or global ecological carrying capacity, dictates that they be treated as critical natural capital (Gardiner, 1994).

In New Zealand wetlands have always played an important role within Maori communities. The relationship is intricate and interconnected, linking the wetland with the people in more than just the material sense (Cromarty and Scott, 1995). Wetlands have provided food, plants for weaving, medicines, dyes, canoe landing sites, places to season timber, and as a store for taonga (Cromarty and Scott, 1995). Yet in the last one hundred years many characteristic New Zealand wetland types have been eliminated, while very few examples are left of others (e.g. kahikatea swamp forest, flax swamp and salt marsh) (Cromarty and Scott, 1995). Indeed, an estimate of the present wetland resource remaining in New Zealand from the New Zealand Land Resource Inventory (Environmental Council, 1983), shows that salt-tolerant wetland types only cover an area of 8,800 ha. This is the lowest coverage from a total of 311,100 ha for all wetland types. Despite our obvious loss, there is little wetland research nor scientific knowledge being applied in New Zealand to ameliorate the situation. The research and application continuing in the United States is much greater, even though the proportion lost is actually less.

There is one hopeful initiative and that is increasing recognition of, and respect for, the rights and traditions of Maori. This brings newly acknowledged responsibilities and gives enrichment to an emerging bicultural perspective on conservation which opens up opportunities for new and innovative approaches in protected area management (Cromarty and Scott, 1995). Admittedly, a bicultural approach is superior, allowing a greater wealth of knowledge to be applied to the problem and providing for continued project support in the form of guardianship (kaitiaki). However, it should not take culture or the grievances of one group to initiate restoration or conservation. It should be a universal commonsense initiative, carried out and maintained because long-term ecosystem functioning depends on it, not to alleviate guilt or provide short-term political popularity.

1.2 Restoration and Creation Rationale

"Ecological restoration is the process of repairing damage caused by humans to the diversity and dynamics of indigenous ecosystems" (Jackson et al., 1995). Ecological restoration is a deliberate intervention that requires carefully set goals and objective evaluation of the success of restoration activities (Jackson et al., 1995). Such criteria might include: the presence, cover and distribution of a plant species; the ability of vegetation to respond to normal disturbance regimes and climatic fluctuations; use by a particular animal species; soil condition and colonisation by particular invertebrates; and characteristics of nutrient cycling and hydrologic regime (Jackson et al., 1995). Rather than controlling nature, restoration is mostly about providing the primary conditions and species that allow a modified area to attain any state within its natural dynamic. This approach is more likely to be sustainable in the longer-term over approaches that attempt to control every aspect of restoration. Such approaches require intensive capital and labour inputs to reach a single specified goal, which may be unattainable under the current environmental regime. If success is demonstrable (either by the criteria above or by restoration of wetland functions) and "results" are seen within human timeframes, public support is likely to be heightened.

However, restoration is not a discrete event. Because the natural world is an evolving and dynamic collection of systems, restoration should be seen as intervention into an ongoing process rather then as a lasting patch or repair (Pickett and Parker, 1994). Currently the "flux of nature" paradigm prevails in ecology, replacing the "balance of nature" idea which dates back to Aristotle and Plato (Aronson et al., 1995). In this new paradigm, natural systems are viewed as open, strongly influenced by processes outside their boundaries, capable of reaching several temporary equilibria, and subject to continual disturbance (Pickett and Parker, 1994). The fact that disturbance or change is not unusual leaves restorationists with many ecological options to consider when planning restoration. It also leads some to believe that as disturbance and invasion are considered natural, and humans are seen as a part of nature and yet another source of disturbance, that restoration is not required. However, the difference with human disturbance is its generality, very high intensity and very high frequency, which can disrupt fluxes in the timeframe of the human mindset. Consequently, Jackson et al. (1995) suggest that when human activity has diminished the flux of nature and when this is predicted to last for more than a human lifespan, then ecological restoration is required. Natural disturbances have similar disruptive timeframes (e.g. landslides and avalanches), but their effects are likely to be more localised and less frequent.

Ecological restoration requires restoration of organisms and their abiotic and biotic interactions. It concentrates on processes such as the persistence of species through natural recruitment and survival; functioning food webs; the integrity of watersheds; and abiotic processes that shape the community such as periodic floods (Jackson et al., 1995). By working with the full diversity and dynamics of ecosystems, sustainable relationships can be restored between nature and culture (Jackson et al., 1995). As pressure on wetlands continues, importing worldwide examples of best practice is needed given the urgency of the situation (Gardiner, 1994).

Pickett and Parker (1994) point out that there is never only one ecologically legitimate state to be used as a blueprint for wetland restoration or creation. Indeed, all ecological systems are influenced by the matrix provided by their surrounding landscape, and by the influences imposed by their unique past, specific spatial setting and current influences (Pickett and Parker, 1994). However, it is desirable to establish at the outset a standard by which comparisons can be made and project design evaluated. Aronson et al. (1995) believe that restoration should be undertaken as an experiment wherein something is defined as a reference, relevant ecosystem attributes are identified, copious baseline data are gathered at the outset, data are collected throughout the project, and enough flexibility is maintained throughout ensuring that the reference ecosystem is a valuable tool rather than a stumbling block (Aronson et al., 1995). If no reference or control is selected, experiments cannot be evaluated (Aronson et al., 1995). Most disciplines tend to fragment and narrow as they develop. This is dangerous in ecological restoration because the systems are complex, and the variety of processes and diversity of linkages so great. Keeping a wide-ranging vision and interacting with people who are focussed on other aspects of ecological systems is a healthy strategy (Pickett and Parker, 1994).

Coastal marshes are particularly dynamic systems and, if it were possible to restore completely, could only be restored to one stage in their frequently changing past. Therefore the aim of restoration should simply be to give coastal marshes the opportunity to attain any state within their natural dynamic.

1.3 The Salt Marsh - A Definition.

Maritime salt marshes occur throughout the temperate world and comprise areas of land bordering the sea, more or less covered with vegetation and subject to periodic inundation by the tide (Chapman, 1974). Originating as bare mud, salt marshes exist wherever the accumulation of sediments is equal to, or greater than, the rate of land subsidence and where there is adequate protection from destructive waves and storms (Chapman, 1974). Thus, the Canterbury salt marshes are predominantly extensions of estuaries, resulting from the interaction between aquatic and terrestrial ecosystems, but with mechanisms that do not exist in either of the adjacent ecosystems.

The ecological structure and function is primarily determined by the tidal regime, soil salinity and nutrient status. To survive in a salt marsh a species must withstand frequent inundations in seawater, soils that are often waterlogged, and mechanical damage by waves (Long and Mason, 1983). Despite these physical restrictions, salt marsh plants are some of the most widely distributed species in the world, dispersed by the transporting powers of the sea, migratory birds and even ships (Ranwell, 1972).

Salt marshes are extremely productive (Long and Mason, 1983; Maltby, 1988; Ursin, 1972). Production is enhanced by the tidal energy which subsidises the solar energy available to plants growing in salt marshes (Odum et al., 1974). Below-ground production and anaerobic decomposition are major processes in the overall energy balance (Pomeroy and Wiegert, 1981). Indeed, as fermentation systems, over a third of primary production is transformed anaerobically (Pomeroy and Wiegert, 1981). Sulphur transformations assume much of the role of oxygen so that a major portion of the below-ground flow of energy cycles through reduced sulphur compounds (Howarth and Teal, 1980).

1.4 Salt Marsh Values and Functions

The value of salt marshes extends well beyond their boundaries. Upland areas are protected from coastal erosion by salt marshes, and food chains both aquatic and terrestrial are fuelled by salt marsh production. At the same time salt marshes are under threat from pollution and reclamation from both agriculture and industry. The marshes of the Avon-Heathcote Estuary are no exception. The estuary is a residual fragment of the sea that once separated Banks Island from the mountains of the South Island (Morgans, 1969). Greatly accelerated by man's influence, the process of land reclamation is continuing in this area.

Salt marshes should be valued as sources, sinks and transformers of a multitude of chemical and biological materials - certainly they are climate stabilisers on a global scale (Mitsch and Gooselink, 1993). For centuries salt marshes in northern Europe and the British Isles were used for shellfishing, grazing, hay production, fences, and thatching for roofs. Today reeds from wetlands are still used for fencing and thatching in Romania, Iraq, Japan, and China (Mitsch and Gooselink, 1993; Queen, 1977).

Salt marsh attributes may be divided into three categories (adapted from Niering, 1985; Broom, 1990; Erwin, 1990; Mitsch and Gooselink, 1993):

1. **Fish and wildlife values**

Salt marshes are essential as habitat for waterfowl, invertebrates and also nursery and spawning grounds for fish and shellfish. Saltwater species tend to spawn offshore, moving into the coastal marshes during their juvenile stages, then migrating offshore as they mature.

2. **Environmental quality values**

Salt marshes enhance water quality, aquatic productivity and microclimate regulation. Water quality is maintained through filtering pollutants, sediment removal, oxygen production, nutrient cycling and chemical and nutrient absorption. Indeed, wetlands have several attributes that cause them to have major influences on chemical materials that flow through them:

- A reduction in velocity of streams entering wetlands causes sediments and chemicals to drop into the wetland. Such nutrients are often stored when they are abundant and released when they are most needed.

- A variety of anaerobic processes such as denitrification remove chemicals from the water decreasing the nutrient input into coastal waters and the likelihood of eutrophication. This is important where inflows are nutrient-enriched such as those from the Bromley Oxidation Pond discharge. However, nutrient input is still required at moderate levels if high fish production is to be maintained.

- The high rate of productivity of many wetlands can lead to high rates of mineral uptake by vegetation and subsequent burial in sediments when the plants die. The high productivity of wetlands is related to the efficient functioning of both the grazing and detritus food chains. There are two major energy flow patterns: (i) the grazing food chain, which mediates direct consumption of green plants; and (ii) the detrital food chain, composed of those organisms that depend primarily on detritus or organic debris as their food source.

- A high water to surface area ratio, because of the shallow depths, leads to significant sediment-water exchange which facilitates the export of production to other areas.

- Lastly, one often overlooked function of wetlands is the removal of CO_2 from the atmosphere. This is achieved by the accumulation of organic carbon in saturated soils that inhibit decomposition (Armentano, 1980).

3. **Socio-economic and cultural values.**

Salt marshes can function to provide: protection from wave damage, erosion control, natural products, fishing and shellfishing, recreation, pleasing aesthetics, education and scientific research areas.

- Fishing and shellfishing: the creation of a salt marsh would provide an extension to the Avon-Heathcote Estuary. The estuary provides a habitat and nursery for populations of both commercial and threatened fish species. For example; flounder, yellow eyed mullet and kahawai. In addition, Eldon and Kelly (1992) refer to the shrinking of inland wetted areas in Christchurch, stating that all major channels appear to be drying up, partly as an unintentional consequence of urban development. This has meant the loss of spawning and rearing habitats that could in part, be provided for in a salt marsh.

- Shoreline anchoring: salt marshes bind shoreline sediments with their root systems, thus anchoring the substrate. The above-ground biomass provides friction against overland sheetflow, wave energy, and storm surges, providing a degree of stabilisation to the shoreline under natural conditions. Rosen (1980) conducted an extensive study of erosion of the Virginia Chesapeake shoreline and concluded that the presence of a salt marsh in the structure of the shore results in the least erosion-susceptible environment.

- Natural products: the financial valuation of wetland functions and natural products is advancing rapidly. Barbier et al. (1993) have shown that the multiple products of the Nigerian Hadejia-Nguru wetlands that can be easily valued (agricultural crops, grazing and fuelwood), and are around 6 times more valuable per hectare than cropping within the 14,000 ha Kano River Irrigation Project.

- Education and scientific research: salt marshes can act as outdoor laboratories demonstrating such basic ecological principles as energy flow, recycling, and limited carrying capacity (Niering, 1985). They can also serve as complex sites for more in depth scientific study.

- Recreation and aesthetics: in times of urban expansion, natural areas within city limits are increasingly valued for their aesthetic appeal and recreation potential. Salt marshes provide for walking, bird watching and picnicking to name a few activities. In addition, studies have shown that people are willing to pay for the privilege of viewing rare species in the wild (Miller et al., 1994).

- Bird breeding and sanctuaries: marsh birds (e.g. bitterns, crakes, rails and fernbirds) are uncommon in the Canterbury region and New Zealand as a whole. If reintroduced, creation of a salt marsh would provide permanent habitat for these birds which dwell almost entirely in dense thickets of aquatic emergent vegetation and in rush a shrub associations on waterlogged soils (Water and Soil Division, 1982), a habitat type sparse in Canterbury. Other coastal and wading birds would be expected ephemerally in the marsh as part of migration, or permanently – using the marsh as an extension of the estuary.

How these socio-economic and cultural values are perceived by individuals will result from the environment and experience (Lasswell, 1971). These values are reflected in attitudes towards nature after being filtered through a cultural context, and may not be rational or reflective of the situation as a whole. The resulting attitudes are translated into specific actions by personal perceptions of the resource, accessibility of the resource, and the personal capacity to act (Casagrande, 1996). It is this individual value process that determines whether a person fishes in, swims in, pollutes or values an ecosystem such as a salt marsh at all.

After surveying residents bordering the West River and associated wetland areas in Connecticut, Casagrande (1996) determined that highest value was placed on aesthetics, the quality of habitat for wildlife, and the ability to relax and see wildlife.

Respondents highly valued passive activities, including walking, relaxing, and enjoying views. They tended to place lower value on active uses, including fishing and boating. Restoration of a salt marsh, by installing self-regulating tide gates that allowed for tidal flushing while providing flood protection, was welcomed by the community (Casagrande, 1996). Here, the tall and impenetrable reed was replaced by short salt marsh grasses creating a landscape with greater visibility and accessibility (Casagrande, 1996). Enhanced visibility increased wildlife viewing opportunities and served to discourage perceptions of danger due to crime (Casagrande, 1996). Preferences for short grass landscapes with high visibility are well documented and may be a result of human evolution (Ulrich, 1993). In New Zealand the mown lawn surrounding most residential housing is evidence of this. However, New Zealanders are also accustomed to dense bush of a range of heights and would probably prefer a salt marsh with a range of species, including coastal trees and shrubs, not just turf-forming marsh herbs.

Restoration would also increase the density of charismatic wildlife. Urban values and knowledge of nature have been studied by Kellert (1984), who suggested that urban residents are likely to believe that wild animals and plants have a right to exist, but that they are also likely to value them as an extractable resource. However, regardless of culture, gender or age, urban residents all preferred natural rather than obviously man-made landscapes (Ulrich, 1993). Natural settings in urban landscapes also have been shown to have psycho-physiologically restorative effects (Ulrich, 1993).

1.5 Wetland Legislation

In addition to the compelling environmental and socio-economic factors, there are a number of national acts and treaties and international agreements and treaties that oblige coastal wetland restoration and creation.

The Ramsar Convention

The Ramsar Convention came into force in late 1975. This global treaty provides the framework for the international protection of wetlands as habitats for migratory fauna that do not observe international borders and for the benefit of human populations dependent on wetlands (Mitsch and Gosselink, 1993). As of 1 July 1986, the convention listed 40 contracting parties throughout the world, including New Zealand. The broad objectives of the convention are to stem the loss of wetlands and ensure their conservation in view of their importance for ecological processes as well as for their rich fauna and flora (Navid, 1988). The convention provides for general obligations relating to the conservation of wetlands throughout the territory of the contracting parties and for special obligations pertaining to those wetlands that have been listed in a 'List of Wetlands of International Importance'. The Avon-Heathcote Estuary is such a listed wetland.

In addition, Article 2 provides that wetlands covered 'may incorporate riparian and coastal zones adjacent to the wetlands'. Contracting parties are obliged to promote the conservation of wetlands in their territory through the establishment of nature reserves.

Agenda 21

Agenda 21 (UNCED, 1992) asserted that "the right to development must be fulfilled so as to equitably meet developmental and environmental needs of present and future generations". The loss of wetlands may be considered as a reduction in the planet's natural capital. Moreover, "some or all (wetland) functions can be restored where they have been lost, or enhanced through rehabilitation measures in degraded wetlands" (UNCED, 1992). It concludes that "the holistic management of freshwater as a finite and vulnerable resource, and the integration of sectoral water plans and programs within the framework of national economic and social policy are of permanent importance....coastal area management and development (requires) new approaches that are integrated in content and are precautionary and anticipatory in ambit".

The Resource Management Act (1991)

In New Zealand, the Resource Management Act 1991 is driven by the principle of sustainability, and this applies to wetland areas as it does to other natural resources. Section 6 of the Act identifies the consideration of wetlands as a matter of national importance. This must be taken into account when powers are being exercised and decisions are being made under the Act. These matters of national importance are:

(a) *The preservation of the natural character of the coastal environment (including the coastal marine area), wetlands, and lakes and rivers and their margins from inappropriate subdivision, use and development.*
(b) *The protection of outstanding natural features and landscapes from inappropriate subdivision, use and development.*
(c) *The protection of areas of significant indigenous vegetation and significant habitats of indigenous fauna.*
(d) *The maintenance and enhancement of public access to and along the coastal marine area, lakes and rivers.*
(e) *The relationship of Maori and their-culture and traditions with their ancestral lands, water, sites waahi tapu, and other taonga.*

The Act also lists "additional matters" which must be taken into account although they are of less significance than "national importance". Wetlands are covered in a general way by such provisions. These include protection of the intrinsic value of ecosystems, the protection of heritage and amenity values, the maintenance and enhancement of the natural quality of the environment, the recognition of any finite characteristic of natural and physical resources, and the protection of trout and salmon spawning habitat. The Act also provides for wetlands to be covered by water conservation orders (Cromarty and Scott, 1995).

New Zealand Wetlands Management Policy

The New Zealand Wetlands Management Policy was adopted by the New Zealand Government in 1986. The objectives of the policy include: the preservation and protection of important wetlands (particularly those of international, national and representative importance); the maintenance of an inventory of wetlands; and the promotion of public awareness of wetland values. The Department of Conservation and City and Regional Councils are responsible for fulfilling these objectives within each region.

1.6 Controlling Authorities

Department of Conservation and local authority policies are consistent with the Resource Management Act and should promote the objectives of the Ramsar Convention and Agenda 21. Such policies provide the context within which development of the Linwood Paddocks can proceed.

The Department of Conservation

The Department of Conservation produced the 'New Zealand Coastal Policy Statement (1994)'. It establishes a national policy for the management and use of the New Zealand coastline. In addition to augmenting the Resource Management Act's policies on sustainability - avoiding and remedying actual or potential adverse effects on coastal areas - it contains the following policies:

1.1.5 *It is a national priority to restore and rehabilitate the natural character of the coastal environment where appropriate.*

3.4.2 *Policy statements and plans should recognise the possibility of a rise in sea level, and should identify areas which would as a consequence be subject to erosion or inundation. Natural systems which are a natural defense to erosion and/inundation should be identified and their integrity protected.*

3.4.3 *In relation to future subdivision, use and development, policy statements and plans should recognise that some natural features may migrate inland as the result of dynamic coastal processes (including sea level rise).*

5.1.6 *Consideration should be given to reducing contamination of natural water in the coastal environment from non-point sources.*

The policy attempts to encourage planning for future environmental change, with particular regard to sea level rise. It contains specific policies which state that restoration of coastal systems is a national priority, especially if such systems are a natural defense to erosion. Hence, salt marsh restoration should be viewed as a national priority.

The Canterbury Regional Council

The Canterbury Regional Council is responsible for the 'Proposed Regional Coastal Environment Plan (1994)'. This requires a resource consent for all discharges into the Avon-Heathcote Estuary including stormwater and effluent. They identify the Avon-Heathcote Estuary as an 'Area of Significant Conservation Value' (Heremaia, 1995). Current discharge consents (including those from the Bromley Oxidation Ponds), are due for renewal by the year 2000. Most current discharges will require additional purification if resource consents are to be renewed.

The Christchurch City Council

In 1995 the Christchurch City Council initiated the Green Edge Proposal. This is coordinated by Christine Heremaia of the Waste Management Unit and involves linking and restoring coastal habitats in Christchurch. It calls for restoration of tidal wetlands on the Linwood Paddocks. Such restoration is consistent with the new (August, 1998) management structure for the 370 kms of waterways and more than 50 wetlands in Christchurch. The new structure takes into account values such as ecology, culture, landscape, heritage and recreation, instead of concentrating solely on drainage (Ken Couling, pers. comm).

However, the Green Edge proposal is currently on hold while the Council debates its City Plan. The new City Plan incorporates the policies pertaining to the natural environment, the city identity, urban growth, recreation and open space. Policies seek to: avoid urbanisation of land which detracts from the margins of waterways; minimise adverse effects upon the values of Tangata Whenua; manage coastal margins to improve the water quality of the Avon-Heathcote Estuary; promote environmental enhancement and rehabilitation of natural areas; enhance the recreation and tourism value of the coastline; and ensure that activities are compatible with the dominant natural values of significant natural areas.

The zoning of the Linwood Paddocks in the draft proposed City Plan (Conservation 1B and Open Space 2) provides for wastewater treatment, significant wildlife values and large areas of public open space for active recreation (Heremaia, 1995).

1.7 Research Objectives and Scope of Study

Research Objectives

- Determination of the historic salt marsh vegetation composition and distribution in Canterbury.
- Determination of the current salt marsh vegetation distribution in Canterbury with respect to elevation, pH, salinity and sediment type.
- Determination of salt marsh vegetation tolerance to natural levels of heavy metal and nutrient levels and also those found in the Linwood Paddocks.
- Assessment of local and overseas restoration projects in an attempt to provide improved restoration designs and management policies.
- Determination of how freshwater wetlands can be designed and which plant species should be used in New Zealand to improve water quality.
- To use the above information to establish goals, design and management criteria that will allow sustainable salt marsh restoration of the Linwood Paddocks.

Scope of Study

To restore the Linwood Paddocks from their current pastoral use to a functioning salt marsh system requires detailed knowledge and scientific understanding of marsh vegetation, hydrology, soils, wildlife, water quality, engineering and economic constraints. Given the thesis duration it is impossible to deal with all aspects in depth. However, simply having a list of plant species is inadequate for implementation and management. Research and experimentation is carried out at a spatial and temporal scale, which should allow sufficient prediction of salt marsh behaviour and a successful restoration design (Chapter 7).

Salt marsh development in the Linwood Paddocks needs to consider specifically: the location adjacent to the Avon-Heathcote Estuary and the Bromley Oxidation Ponds; the elevated nutrient and heavy metal level as a result of past biosolid application; and the threat of sea level rise (Chapter 2).

The survey (Chapter 3), experimental (Chapter 4) and assessment (Chapter 5) component of this thesis is focussed on restoring the vegetation structure and composition that best enables the system to attain a salt marsh state. Restoration of the primary producers and hydrological links should facilitate the introduction of additional species and allow for a diversity of functions. Because the system has been allowed to obtain any state within a natural salt marsh dynamic, labour and capital inputs after the initial revegetation should be much reduced.

Creation of freshwater wetlands to improve the water quality of the potential Bromley Oxidation Pond freshwater source, is a further option that would sustain salt marsh restoration and enhance salt marsh and estuary wildlife and water quality values simultaneously (Chapter 6). In addition to providing a prescription for restoration that increases the habitat value for wildlife, there are also education and tourism opportunities to be realised (Chapter 7).

2. THE LINWOOD PADDOCKS

2.1 Introduction

The area known as the Linwood Paddocks is Christchurch City Council-owned land which lies between the Bromley Sewage Treatment Works and the Avon-Heathcote estuary. Some of the surrounding land is zoned as residential and further residential developments are expected in this area (Heremaia, 1995). It is therefore imperative that Council-owned land (Linwood Paddocks) along the estuary margins is retained in order to act as a buffer or corridor, or extended wildlife habitat, between existing and proposed developments (Heremaia, 1995). Certainly, the areal extent and location (Fig. 2.1) of the Linwood Paddocks provide the Council with a unique opportunity to enhance estuary wildlife and water quality values simultaneously.

Historical development of the Linwood Paddocks has proceeded like most development within New Zealand, with little regard for continuing environmental or heritage values. Although somewhat modified, the Linwood Paddocks have the potential to become an important part of the Christchurch City Council's Green Edge Proposal. The Green Edge Proposal is part of an initiative designed to link up fragmented vegetation and wildlife habitat within Christchurch and to provide a continuous vegetated edge along the coastline (Heremaia, 1995).

The Avon-Heathcote Estuary is classified as having very high ecological values. Certainly, it is the largest enclosed estuary within the Canterbury Region and it is one of the most important coastal wetlands in New Zealand, particularly for migratory shorebirds. The Estuary is also of special value in maintaining the genetic and ecological diversity of the region because of its relatively large size and high species richness (e.g. 34 species of fish representative of both marine and freshwater habitats). In order to maintain these values, consideration must also be given to the adjacent land, because all are interlinked hydrologically and through wildlife use. Despite modification and primary use for farming, the Linwood Paddocks still provide an important buffer to the Estuary as well as habitat for pukeko and some coastal bird species (Heremaia, 1995). Until recently, the paddocks were utilised for biosolid recycling. Currently they are only used for cattle grazing. With restoration to a more natural and diverse system, the paddocks could provide habitat for a wider range of species (e.g. aquatic invertebrates, fish and marsh birds) and act as a more substantial buffer to the estuary by performing additional functions including water purification.

Sustainable management of the natural environment is the aim of current legislation and the national, regional and city plans (previously outlined) that govern development of the council-owned Linwood Paddocks. Apart from legislation, the main restrictions governing use of the Linwood Paddocks concern sea level rise and heavy metal level. This chapter will focus on the historical situation of the Paddocks and surrounding areas, and the future issues and options of development.

2.2 Site Location

The Linwood Paddocks occupy 90 ha of low-lying land adjacent to the Avon-Heathcote Estuary. This area is only 5 kms from Cathedral Square and is situated on a potential scenic route incorporating a number of tourist attractions (e.g. Gondola, Ferrymead, Sumner) (Heremaia, 1995) (Fig. 2.1).

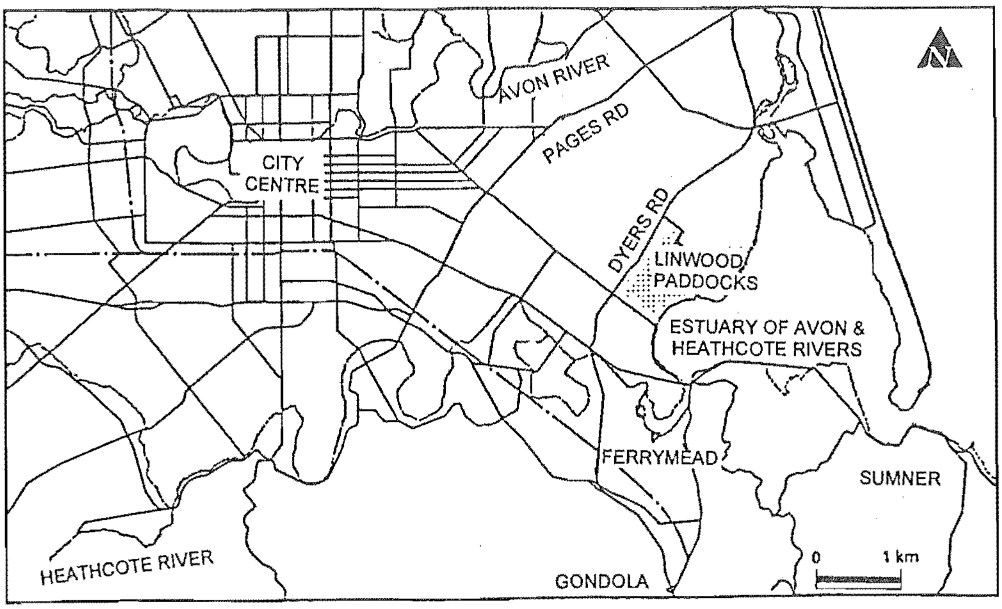


Figure 2.1 Location map of the Linwood Paddocks. The large section of coastline encompassed by the Paddocks means that they could form a substantial buffer between the Estuary and upland urban development.

2.3 Historical Development

Past archaeological discoveries in the immediate vicinity have confirmed that Maori people lived beside the Avon-Heathcote Estuary for several hundred years before the arrival of the first Europeans (Cromarty and Scott, 1995). During pre-European times the Avon-Heathcote Estuary was among the most important and highly valued food-gathering sites for Maori on the East Coast of the South Island (Cromarty and Scott, 1995). The estuary was rich with eels, lamprey, inanga, flounder, shellfish and a snail called "whetiko" (Cromarty and Scott, 1995). A number of features have Maori names, including the Avon River (Otakoro), Heathcote River (Opawaho) and the New Brighton Spit (Te Karoro Karoro).

Europeans, in contrast, viewed the wetlands and sandhills that adjoined the Linwood Paddocks as uneconomic wastelands. This historical perception of the area is reflected in the type of activities that have or still occur in this area today (e.g. sewage and rubbish disposal, industry, and cemeteries) (Heremaia, 1995). Sewage discharge into the estuary has severely compromised the value of the estuary as a food-gathering area for Maori such that the local Ngai Tahu have put a "rahui" (prohibition) on collection of shellfish from the area (Cromarty and Scott, 1995).

Though information is scarce, the Linwood Paddocks were clearly a wetland of some description with swampy areas of raupo further inland prior to drainage c.1882 (Fig. 2.2). The vegetation bordering the Heathcote and Avon River mouths at this time was predominantly rushes and flax, with raupo further inland (Fig. 2.2). Cromarty and Scott (1995) report that plant communities on lands bordering the estuary in the early 1850s were dominated by raupo *Typha orientalis* and New Zealand flax *Phormium tenax*, interspersed with tussock, fern and tutu on drier areas to the south and west, and sea rush *Juncus maritimus* backed by saltmarsh ribbonwood *Plagianthus divaricatus* along the New Brighton Sandspit.

No photos or paintings are available of the Linwood Paddock area for this period. However, there are several of the Heathcote and Avon River mouths close by. The

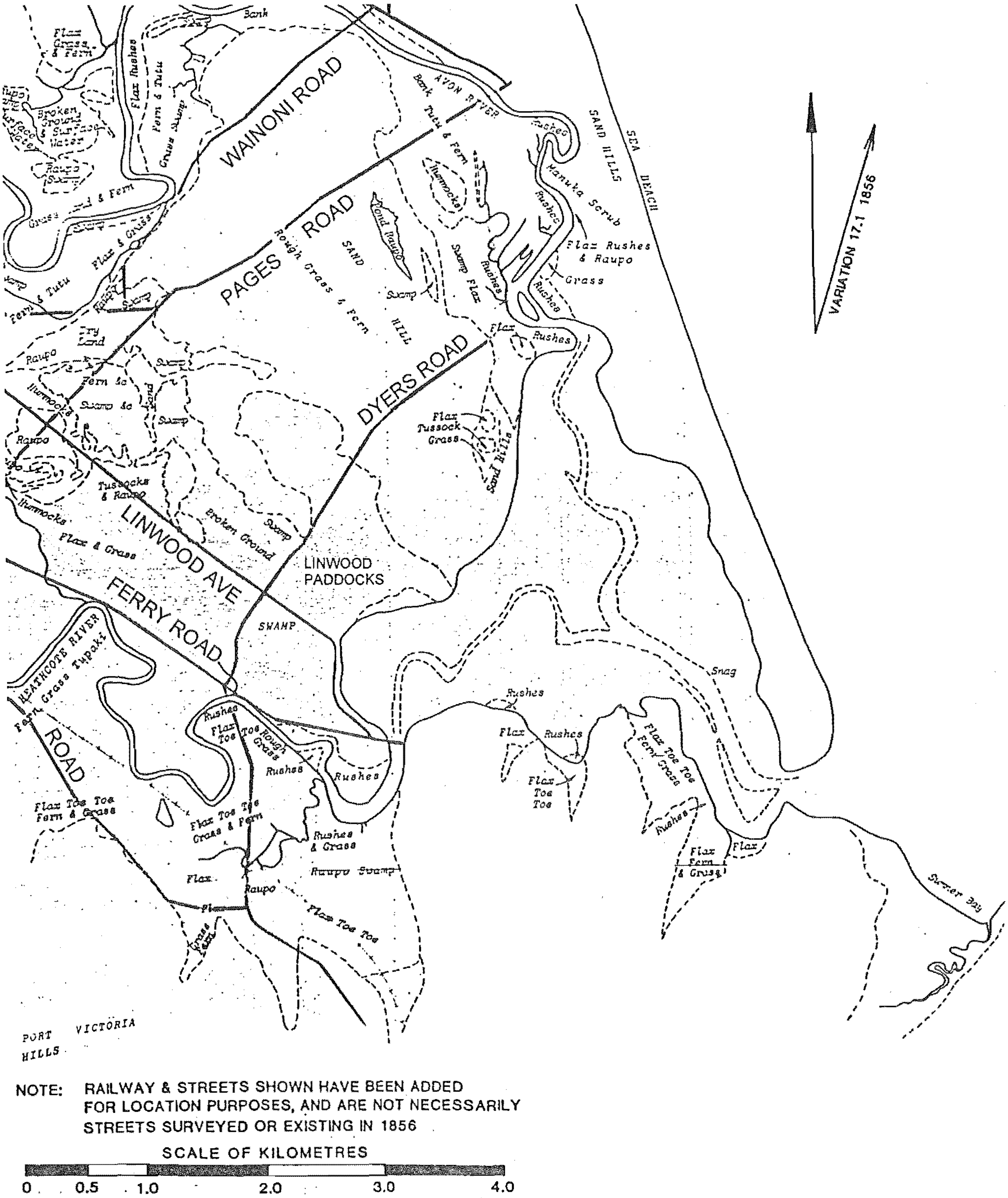


Figure 2.2. The Christchurch area surrounding the Linwood Paddocks showing waterways, swamps and vegetation type in 1856. Compiled from 'Black Maps' approved by J. Thomas and T. Cass, Chief Surveyors 1856. Source: Wilson, 1989.

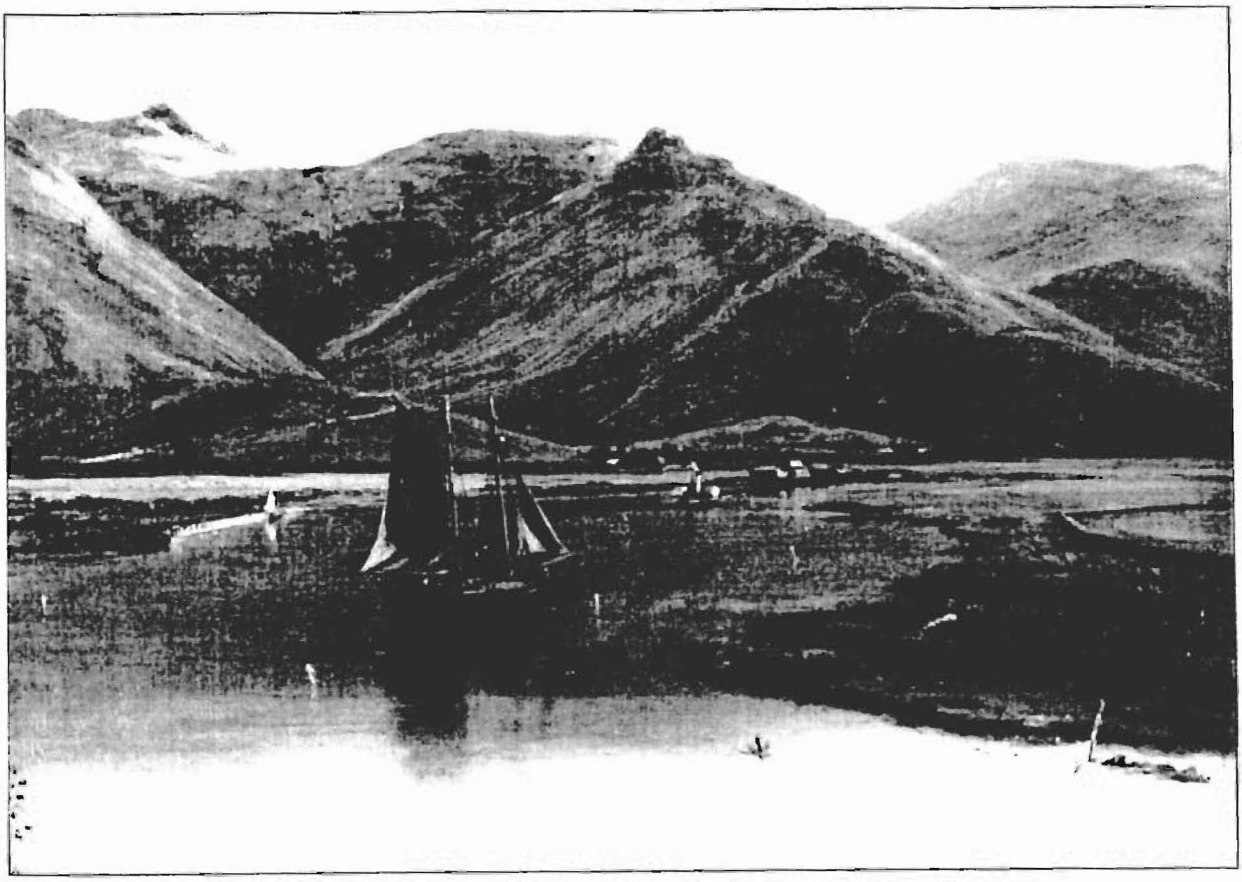


Figure 2.3. The Heathcote River, Ferrymead. Oil painting by John Gibb, 1885 (CML collection).

Figure 2.4. The Heathcote River, c. 1900 (W. A. Taylor collection)

Figure 2.5. The Heathcote River, Woolston, c. 1900 (W. A. Taylor collection).

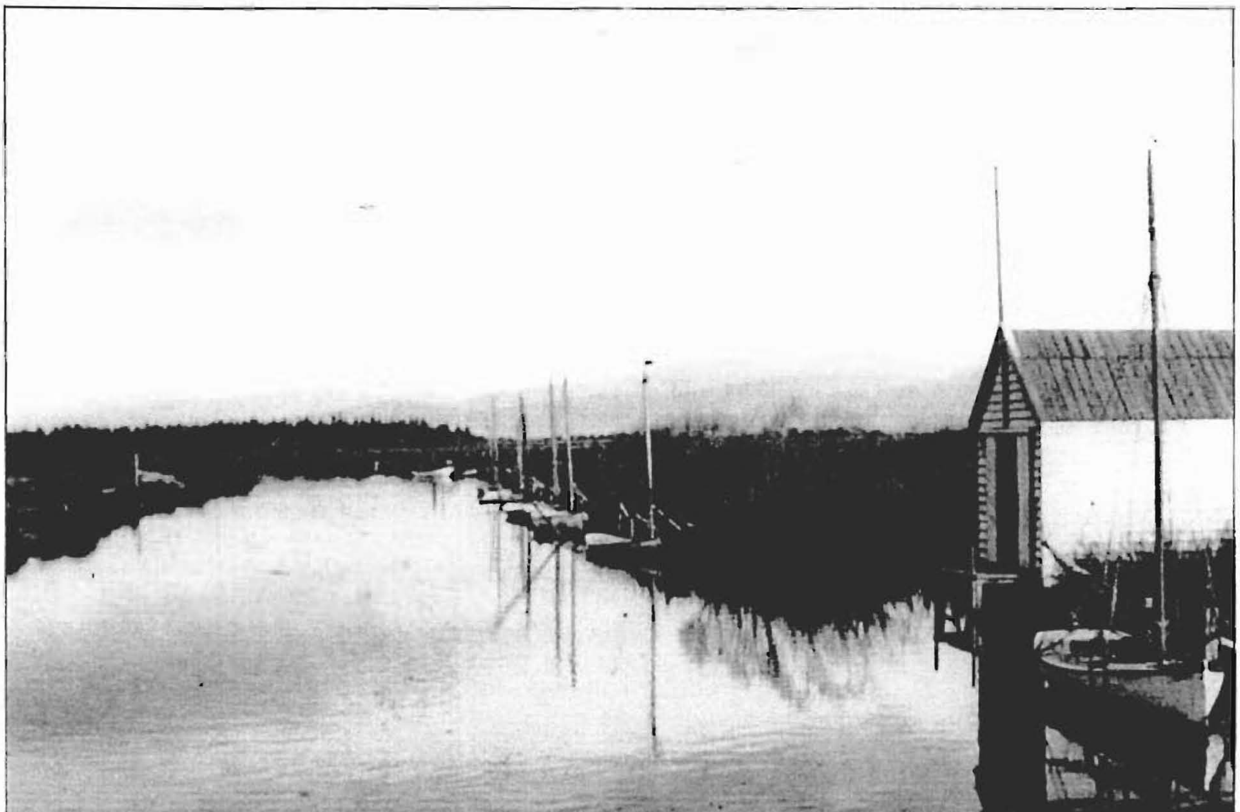


Figure 2.6. The Lower Avon River, c. 1900 (CML collection)

Heathcote River salt marsh consisted of *Juncus maritimus* that extended further out into the channel and was denser than at present (Fig. 2.3). Figs. 2.4 and 2.5 also show the Heathcote River in 1900. New Zealand flax, *Phormium tenax* is clearly visible, growing right up to the water's edge in both photos. Knobby clubrush, *Scirpoides nodosa,* is also visible in the foreground of Fig. 2.4. The lower Avon River had extensive stands of rushes (Fig. 2.6). Again, the vegetation extends further into the channel than at present, and appears to be denser than the current distribution.

Certainly there is a contrast between wetland extent and landuse prior to European occupation and that remaining today (Figs. 2.7 and 2.8). The remaining salt marsh areas represent only about 10 % of the original, however, the diversity of plant species remains high (McCombs and Partridge, 1992). The most extensive areas of native vegetation today, are the salt marsh communities prominent along the South Brighton Spit and extending to the Avon River mouth (McCombs and Partridge, 1992). Sea rush *Juncus maritimus* and glasswort *Sarcocornia quinqueflora* are the most common species immediately above mid-tide, with *Selliera radicans*, buck's horn plantain *Plantago coronopus*, sea primrose *Samolus repens* and tussock sedge *Carex secta* appearing further up the shore. Plant communities in the vicinity of the Avon River include lake club rush *Schoenoplectus validus,* jointed wire rush *Leptocarpus similis,* New Zealand flax *Phormium tenax,* raupo *Typha orientalis* and saltmarsh ribbonwood *Plagianthus divaricatus*. Communities of smaller plants closer to the shore include native musk *Mimulus repens*, bachelor's button *Cotula coronopifolia,* orache *Atriplex prostrata* and three-square *Schoenoplectus pungens.*

Progressive development of housing, roading, sewage treatment facilities and other amenities by the City of Christchurch has resulted in the destruction or extensive modification of the original native vegetation bordering the estuary, particularly along western and southern shorelines where original plant communities have been replaced by open grassland bordering the Bromley sewage ponds and Linwood Paddocks, or completely destroyed by road construction which skirts the southern shoreline from the Heathcote River to Moncks Bay (Cromarty and Scott, 1995). The Linwood Paddocks were created in 1882-83 when an area of sandhills, some up to 10 m high,

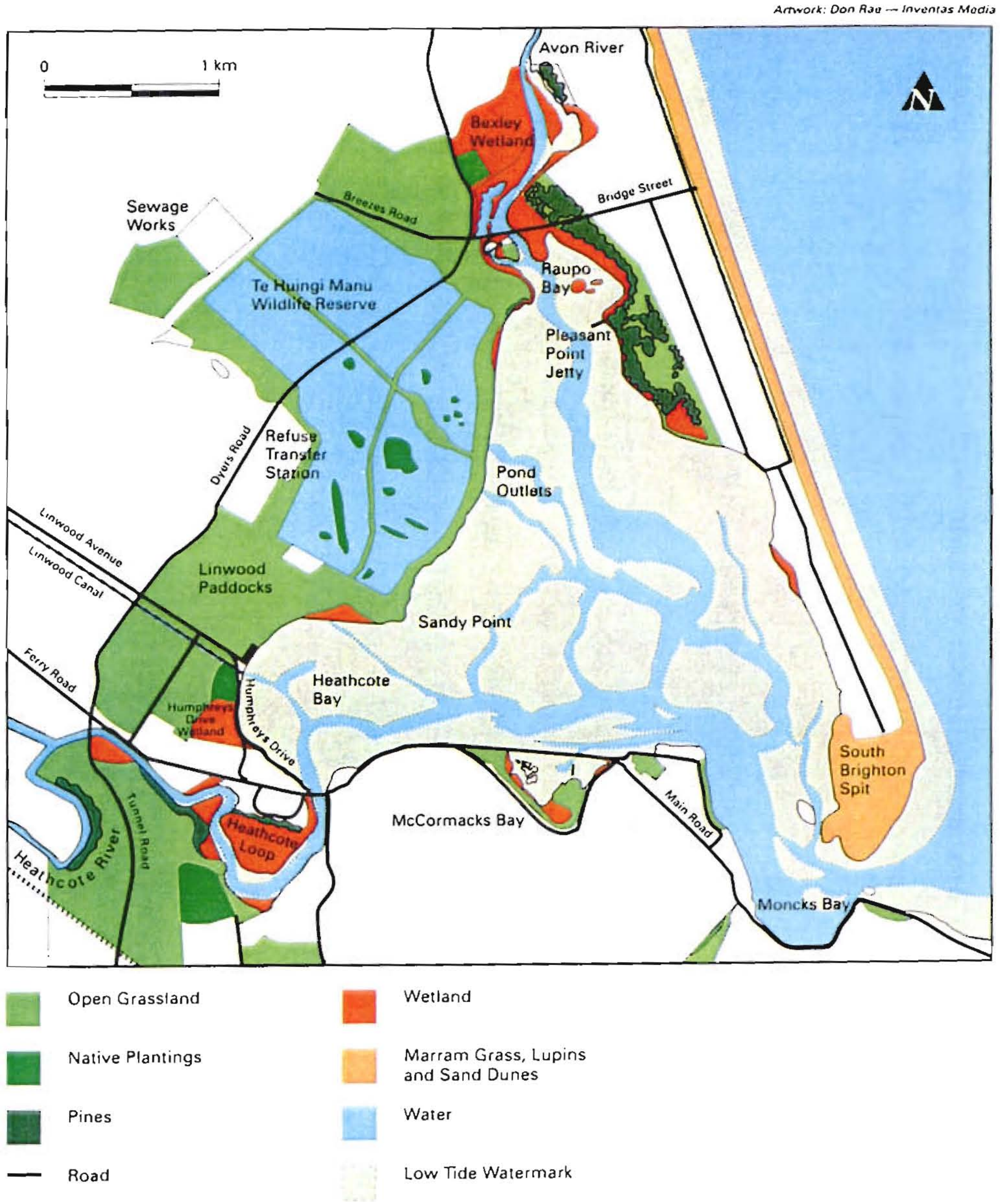


Figure 2.7. The Avon-Heathcote Estuary and remaining wetlands today (adapted from Owen, 1992).

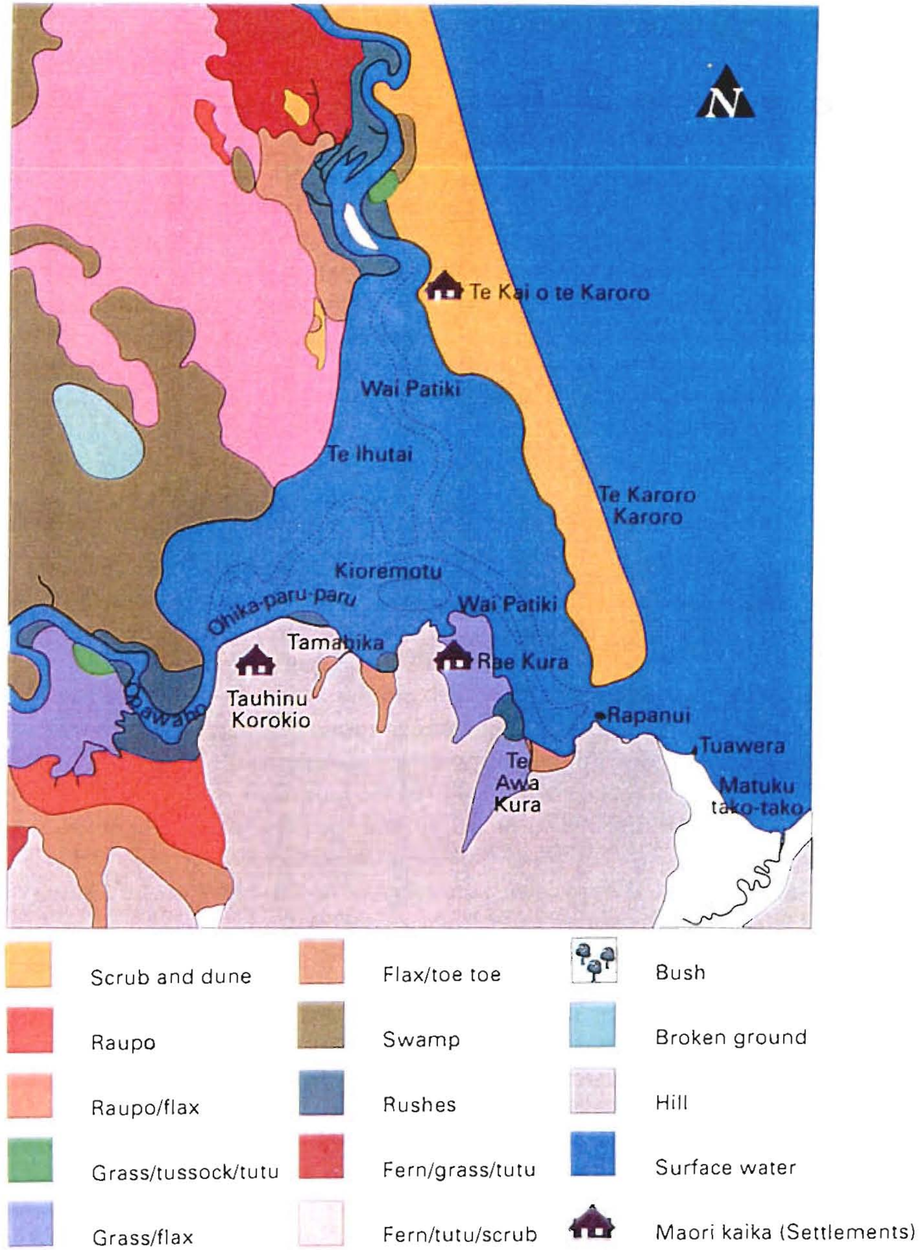


Figure 2.8. The Avon-Heathcote Estuary prior to European occupation (adapted from Owen, 1992).

were flattened (Wilson, 1989). The resulting 19 ha of flat land was divided into nineteen paddocks, and drainage races run between every two paddocks (Wilson, 1989). From settling ponds, sewage was directed down the race system and then distributed through sluices over the Paddocks (Wilson, 1989). After more than twenty-five years, black loam soil supporting pasture species had developed where the wetlands and dunes had once been. In the early 1900s, more sandhills were leveled to add an extra 7.5 ha to the sewage farm (Wilson, 1989). After 1962, the Drainage Board's farming operations continued but changed significantly in character. Land was now used not to filter only partially treated sewage but for the disposal of digested sludge (Wilson, 1989). The Linwood Paddocks are almost entirely introduced pasture species (e.g. ryegrass and clover) and are currently separated from the Estuary by tide gates and a retaining wall. Fattening of steers and stud operations still continue on the farmland today.

2.4 Heavy Metal Considerations

One of the major concerns about utilisation of the Linwood Paddocks for wetland restoration/creation is the level of heavy metals accumulated in the sediments as a result of biosolid application, which ceased in 1995 following Department of Health concerns.

Heavy metals, as defined by Viarengo (1989), are a group of elements with atomic weights ranging from 633.5 to 200.6 and characterised by similar electronic distribution in the external shell (e.g. Cu, Zn, Cd, and Hg). Although these elements are toxic to estuarine and marine organisms above a threshold availability, many of them are essential to metabolism at lower concentrations (e.g. Co, Cu, Fe, and Mn) (Rainbow, 1985; Kadlec and Knight, 1996). Of great concern as biological pollutants are antimony, arsenic, cadmium, chromium, copper, lead, mercury and selenium, all of which have contributed to severe insidious pollution problems in various estuarine and coastal ecosystems (Kennish, 1992; Novotny, 1995). Some heavy metals, such as cadmium and lead, have no known biological function and may negatively affect/poison biotic communities (Kennish, 1992). At elevated levels heavy metals alter the functioning of organisms by acting as enzyme inhibitors (Kennish, 1992).

The responses of estuarine organisms to the toxic effects of heavy metals are manifested in a variety of ways. In general, changes occur in organism physiology, reproduction, and development causing growth inhibition or disfigurement (Kennish, 1992). Because of such persistence in the environment, toxicity at high concentrations, and their tendency to accumulate in the tissues of biota, heavy metals pose potentially hazardous conditions for humans (Kennish, 1992).

Heavy Metal Level in the Linwood Paddocks

As part of their continued monitoring the City Council have tested the Paddocks for a range of heavy metals and nutrients.

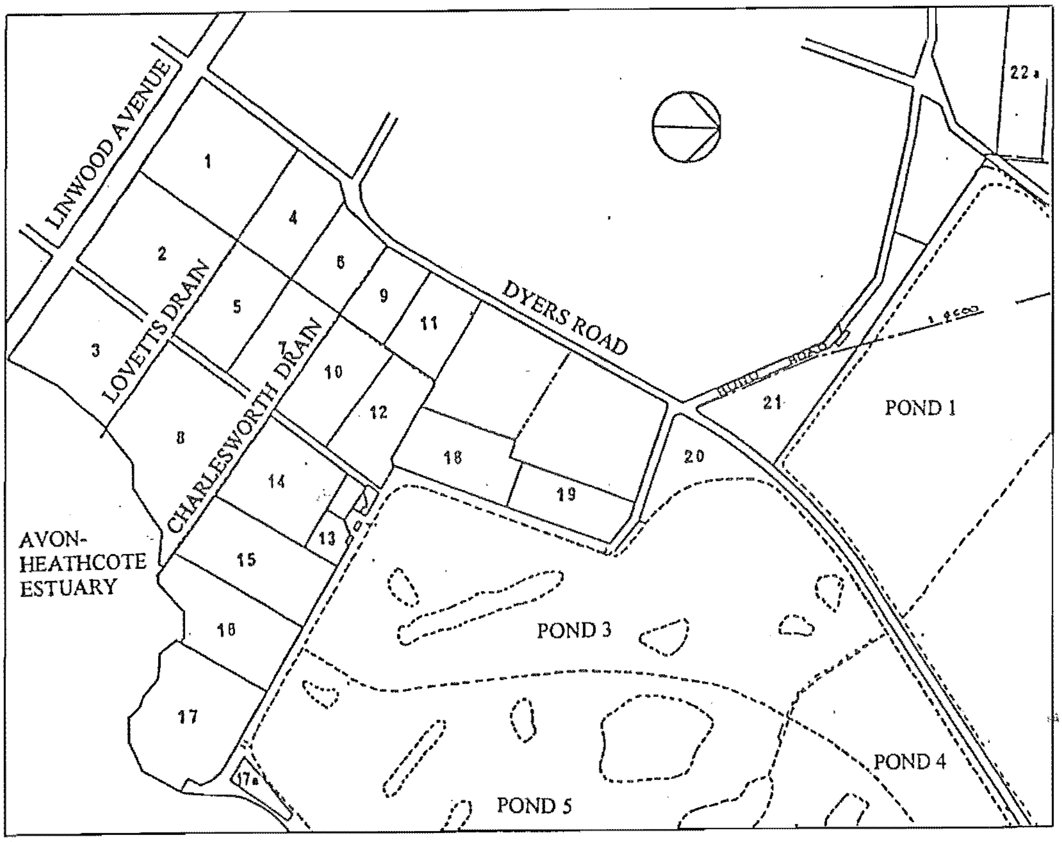


Figure 2.9 Layout of the Linwood Paddocks and the adjacent oxidation ponds.

Table 2.1 lists the levels recorded in the last survey of those Paddocks likely to be utilised for wetland restoration/creation. Samples were taken from the top 15 cm of each paddock.

Table 2.1 Results of a heavy metal and nutrient survey carried out by the City Council 1992-94. All metals are expressed as average mg/kg.

Paddock No.	Cu mg/kg	Cr mg/kg	Ni mg/kg	Zn mg/kg	Cd mg/kg	Pb mg/kg	P mg/kg	K mg/kg	N%
3	75.2	195.0	23.9	204.4	0.6	66.2	43	430	1.2
8	36.1	40.1	15.1	106.0	0.1	35.6	38	490	1.14
13	61.6	256.3	20.9	243.2	0.7	59.7	39	230	0.17
14	46.6	145.4	19.8	203.1	0.2	47.5	61	140	0.26
15	191.1	644.7	46.6	549.5	1.9	138.6	250	130	0.61
16	136.3	536.2	40.3	386.0	1.2	91.1	170	300	0.82
17	391	834.6	63.0	740.4	3.3	218.8	300	230	0.72
17A	168	695	29	572	5.3	123	380	310	.66

In addition to these surveys, analysis on samples collected in May 1998, revealed that all samples comply with the New Zealand Department of Health guidelines, except that for arsenic. The soil from Paddock 8 has an arsenic level of 21.45 mg/kg (Table 2.2) which is more than double the maximium allowable level of 10 mg/kg.

Table 2.2 Results of a heavy metal and nutrient survey carried out in May, 1998.

Paddock No.	Cu mg/kg	Cr mg/kg	Ni mg/kg	Zn mg/kg	Cd mg/kg	Pb mg/kg	As mg/kg	P mg/kg	N mg/kg
8	15.14	13.46	10.52	70.5	0.04	31.34	21.45	1145	5643

Presently the arsenic source is unknown, and further tests are required to determine whether arsenic levels are elevated throughout the Paddocks or whether this is an anomaly. Arsenic is environmentally ubiquitous and elevated levels are generally the result of industrial emissions from nonferrous primary metal smelters, as well as from the mills of some gold mines, both of which process ores containing substantial amounts of arsenic (Blejer, 1975). Agricultural spraying and other applications of arsenical economic poisons, including rodenticides, fungicides, weed-killers, sheep-dips, insecticides and other pesticides are also important sources of arsenic (Blejer, 1975).

Arsenic is not an essential element in human nutrition, physiology or biochemistry (Blejer, 1975). It is a potent, cumulative protoplasmic poison that interferes with sulfhydryl groups in enzymes. Plant species differ greatly in their sensitivity to both heavy metal deficiency and level of toxicity. For arsenic in particular, competition for sites with potential metabolites and occupation of sites for essential groups such as phosphate and nitrate, are thought to be the main mechanisms causing phyto-toxicity (Alloway, 1995). Arsenic can also be accumulated to extreme levels by populations of plants which have recently evolved tolerance on arsenic-toxic substrates (Peterson, 1975).

Unfortunately As, Cu, Ni, Pb, Se, and Zn have residence times of 1000-3000 years (Alloway, 1995), thus their effects will continue to be felt long after their introduction.

Linwood Paddock Restoration Considerations

The main processes of concern when re-flooding the metal-enriched Linwood paddock soils as part of salt marsh restoration are: (i) release of metals to surface water; (ii) metal uptake by wetland plants; (iii) metal accumulation by benthic and wetland animals; (iv) runoff losses; and (v) leaching losses. Fortunately, except where a flooded soil or sediment becomes strongly acid upon drainage and oxidation, heavy metals are more strongly immobilised under pH-neutral, reducing wetland conditions, compared to upland soil conditions (Mikkelsen and Brandon, 1975; Reddy and Patrick, 1977) and large-scale metal releases from contaminated soils and sediment do not occur with changing redox conditions (Gambrell, 1994).

Vascular plants used in revegetation (e.g. *Typha sp.*) may incorporate heavy metals in their tissues. The concentrations of heavy metals in plant parts are the result of uptake and translocation processes (Otte et al., 1993). These are dependent of the availability of metals as determined by the physical and chemical properties of the soil such as soil particle size distribution, pH and redox potential (Salomons and Forstner, 1984). Fortunately, Linwood Paddock soils are predominantly clay (84 %), thus water permeability is much reduced and uptake by plants or leaching of metals (strongly adsorbed by clay colloids, many forming insoluble crystalline compounds whose mobility is relatively low) would be minimal. For example, when plants are growing in substrates with similar arsenic concentrations, lower levels are found in plants grown on clays and silts, with their higher clay mineral and Fe/Al oxide content, than in plants grown on lighter soils (e.g. sands or sandy loams) - in fact arsenic in sandy soils is approximately five times more available with regard to its toxic effects than in clay soils (Alloway, 1995).

Immobilization can be further enhanced in a salt marsh through the use of concretion-forming wetland plant species that can tolerate initial heavy metal levels and become established. Through transport of oxygen to their roots (which are surrounded by anoxic sediment), vascular plants are able to extract metals from the bulk sediment and concentrate them into oxidised microenvironments surrounding each root (Cacador et al., 1996; Sundby et al., 1998). Any vascular marsh plants that have a well developed aerenchyma system and are therefore efficient conduits of oxygen are capable of this. Species that have been studied include *Spartina* spp. and *Aster* spp. (Otte et al., 1993). Doyle and Otte (1997) further believe that burrowing invertebrates (e.g. *Arenicola marina*) in association with vegetation could considerably affect the retention of metals through oxidation of the rhizosphere/burrow wall. The resulting concretions may be produced in the space of a few weeks and are 5 - 10 times enriched in Cd, Cu, Pb, and Zn with respect to the surrounding sediment (Sundby et al., 1998). Rhizoconcretions have been found in preindustrial sediments and areas of saltmarshes that are no longer vegetated (Sundby et al., 1998). Such persistence of concretions upon burial into the anoxic subsurface sediment suggests that metals will remain immobilised once the concretions have formed.

Prevention of heavy metal leaching into the Avon-Heathcote Estuary is important both locally and nationally. A study on the sediment toxicity in the Avon-Heathcote Estuary by Nipper et al. (1997) found that sediment concentrations for lead, copper and zinc already exceeded guideline levels. They concluded that sediment toxicity in the estuary could be sufficient to cause changes to biological community structure and may have flow on effects to higher organisms including fish and birds.

In New Zealand as a whole there has been a continuous increase in heavy metal contamination of estuarine and coastal marine waters. This is directly attributable to industrialization and development in the coastal zone (Kennish, 1992). Heavy metals enter estuaries from three main sources: (i) freshwater input; (ii) the atmosphere; and (iii) anthropogenic activity. The use of fossil fuels, antifouling paints, smelting, power station corrosion products (Cu, Cr, and Zn), seed dressings and slimicides (Hg), automobile emissions (Pb), and other industrial emissions all contribute to anthropogenic input (Kennish, 1992). Sewage-sludge disposal, ash disposal and dredged-material disposal are also major sources. The sources of these pollutants may be industrial or municipal, and may include both historical and current discharges.

Bromley Oxidation Pond discharge (Table 2.3) and urban stormwater runoff (carried by the City Outfall drain and the Avon and Heathcote rivers) are the major anthropogenic sources of heavy metal contaminants in the Avon-Heathcote Estuary. These are also the sources that most likely contribute to toxic heavy metal levels.

Table 2.3. Typical levels of municipal sewage sludge constituents at the final ponds of the Bromley Treatment Works determined from monthly monitoring (Department of Health, 1992).

Element	Cu	Cr	Ni	Zn	Cd	Pb	As	Mg
Level mg/kg	572	1618	117	1648	6.2	443	10.9	5.1

The creation of freshwater wetlands followed by a restored salt marsh is one way of reducing current anthropogenic discharges into the Avon-Heathcote Estuary. Fresh water wetlands could be designed to filter discharge from the adjacent Bromley Oxidation Ponds and supply an adjacent salt marsh with a reliable supply of fresh water. The restored salt marsh would then provide a buffer that filters both pollution sourced from within the Linwood Paddock water shed and any residual from the fresh water wetlands.

2.5 Sea Level Rise

Due to the inherent uncertainty and assumptions surrounding sea level rise, the precise effects are almost impossible to predict. However, it is an extremely important reality, especially in such a low-lying area as the Linwood paddocks.

It is worth noting that historically mean sea levels around New Zealand have been rising. Indeed, sea level rise has been a factor in coastal landuse for at least all of this century, whether or not controlling agencies were aware of it (Oliver and Kirk, 1992).

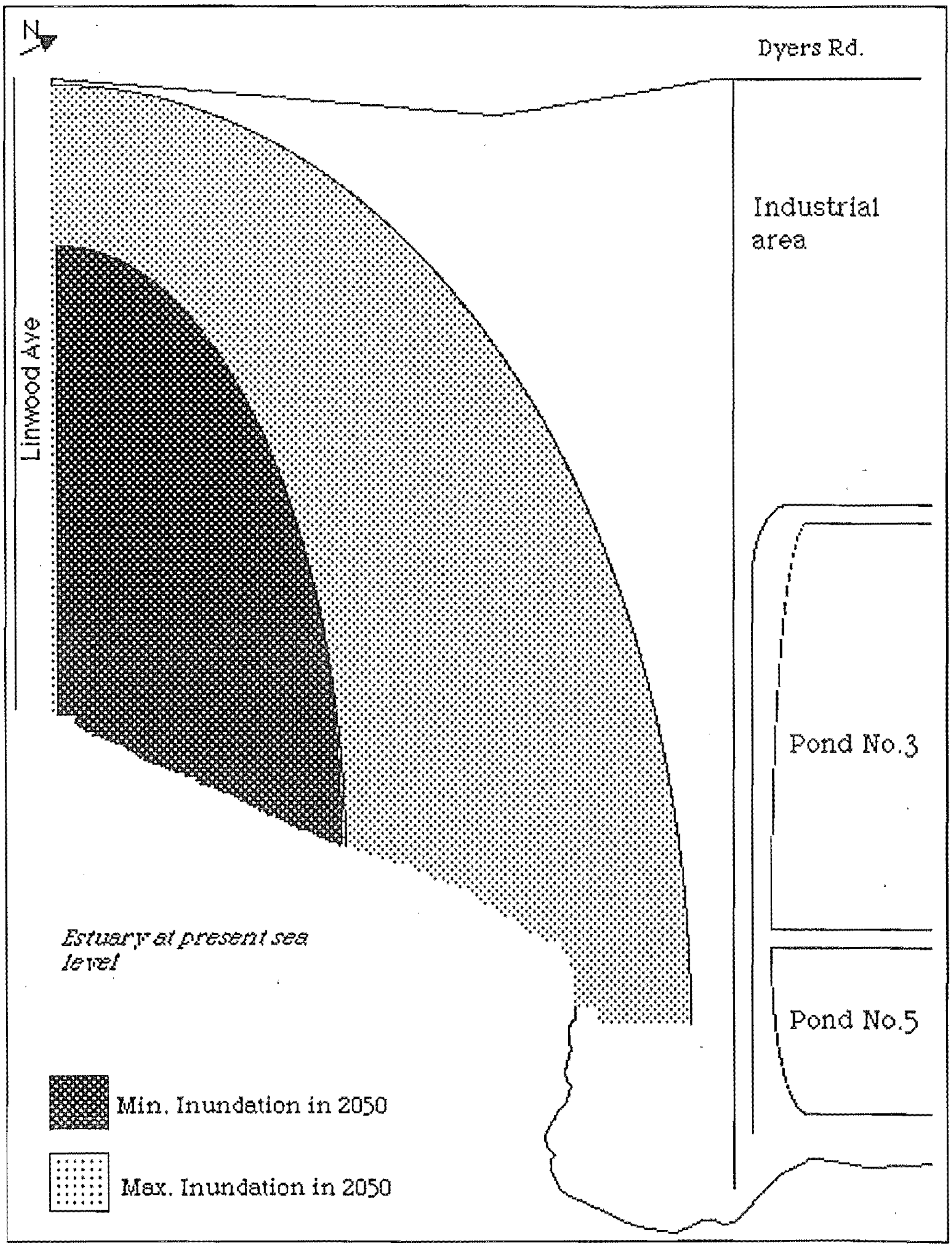


Figure 2.10. Predicted sea level rise for the Linwood Paddocks. Minimum and maximum inundation levels are based on a range of sea level rise predictions recommended for planning purposes for the Christchurch City Council by D. Hamilton (1992). These levels have been added to the 95th percentile of recorded high tides in the Avon-Heathcote Estuary.

Moreover there is no reliable evidence that this rise has any connection to "Greenhouse" warming of the climate and no evidence of any acceleration of the rate of rise during this century (Oliver and Kirk, 1992). It is therefore a minimum necessity to plan to sustain future rises of mean sea level that occur at least at the known historical rates. Fig. 2.10, is a stylised prediction of sea level rise in fifty years time. It is important to note that whilst sea level may be rising, landuse also plays an important role in the resulting inundation. Landuse that accretes sediment (e.g. a salt marsh) may make the effects of sea level rise negligible. Landuse that accelerates erosion or maintains the current level will show increased effects of sea level rise and result in erosion and inundation. Chapter 7 details the processes involved in shoreline erosion resulting from an increase in sea level rise. The specific salt marsh attributes that lessen the negative effects of sea level rise are also outlined in that chapter.

2.6 Conclusion

The loss of both salt and freshwater native wetland vegetation in this area has been substantial. With this loss in habitat, the values and heritage of the local Tangata Whenua are lost also. Salt marsh creation is one way of mitigating this loss. It also creates a link between remaining salt marsh remnants, thus providing for their continued survival.

The close proximity to the Christchurch city centre and location on a potential scenic route make salt marsh creation a realistic tourism and education option. Furthermore, the location adjacent to the Bromley Oxidation Ponds provides the opportunity to create freshwater wetlands that will improve Oxidation Pond water quality and provide a reliable water supply to the marsh restoration.

Sea level rise (although not always obvious) is a real concern and threatens development in this area - not only of the paddocks themselves but of that further inland as well. Development of a salt marsh would lessen the effects of sea level rise. Indeed, a salt marsh results in the least susceptible environment to coastal erosion (Rosen, 1980).

Heavy metals are likely to pose serious threats in such a confined area. Fortunately, it appears that for most metals, application has ceased before accumulation of levels toxic to plant and animal life. The qualities of heavy metals must be assessed in an objective manner. The fact that a metal is persistent, toxic or liable to bioaccumulate, does not necessarily spell disaster if present (Gray and Bewers, 1996). Often it is implied that elements should be banned from discharge because they are intrinsically persistent, or because they are toxic (Gray and Bewers, 1996). It is important to look at the concentration and the species that will likely be affected, if any. In small trace quantities, many metals are necessary for aquatic life and human health (Novotny, 1995). Certainly, life as we know it would not exist if some substances (e.g. zinc and copper) were not persistent or bioaccumulative (Chapman, 1997). However, continued discharge (resulting in a high concentration of heavy metals, especially Pb, Cu and Zn) into the Avon-Heathcote Estuary (a localised area with a range of organisms) is likely to be hazardous. This may be practically ameliorated only by reduction at source, and treatment prior to discharge - including retention in a wetland.

Within limits, creation of a salt marsh in this area would immobilise heavy metals through adsorption onto clay particles, retention as insoluble sulfides under neutral pH conditions, and uptake and immobilisation (formation of persistent rhizoconcretions) by vascular wetland plants. Salt marsh restoration would also provide more wildlife habitat and enhance the quality and persistence of that remaining. Currently, the destruction of habitats and 'reclamation' of wetland and coastal areas, rather than pollution, are likely to cause the most pervasive effects on marine biota.

3. REFERENCE SALT MARSHES

" Who can explain why one species ranges widely and is very numerous, and why another allied species has a narrow range and is rare? Yet these relations are of the highest importance, and they determine the present welfare, and, as I believe, the future success and modifications of every inhabitant of this world" (Darwin, 1859).

3.1 Introduction

In New Zealand, salt marshes are the principle ecosystem in the wetland coastal environment from Tauranga southwards (Chapman, 1977). Like plants in every habitat, salt marsh plants are important because they are the primary producers and the beginning of all complex food webs. Salt marsh species also act as sediment stabilisers and provide structural habitat in this extreme environment.

In salt marsh communities most plant species are clonal perennials, which spread within marshes almost entirely by vegetative growth. The resulting plant zonation is conspicuous and has attracted the attention of ecologists for over a century (e.g. Shaler, 1886).

Where a large number of environmental factors operate, it is often difficult to determine which factors actually determine vegetation patterns (Partridge and Wilson, 1989). In the case of salt marshes, the ultimate cause is the interaction between the tide and elevation. The development and zonation of vegetation in the salt marsh is further influenced by several proximal chemical factors all affected by the tidal regime. The most important of these are the soil water salinity and the degree of anaerobiosis, which controls the pathway of decomposition and nutrient availability. Although marsh plants grow in full sunlight on nutrient-rich sedimentary material, and appear to have a limitless water supply, production rates are variable (Mitsch and Gooselink, 1993). The concentration of dissolved salt means that although physically plants have enough water, physiochemically they are lacking, consequently, plants must expend energy to increase their internal osmotic concentration in order to take up water. As a result, numerous studies confirm that plant growth is progressively inhibited by increasing salt concentrations in the soil. This is true even for the most salt-tolerant species of the salt marsh (Pomeroy and Wiegert, 1981; Mitsch and Gooselink, 1993).

Recent research has revealed the importance of biotic interactions, particularly competition, in plant zonation (Levine et al., 1998; Sanchez et al., 1998). This has lead to a new paradigm for the zonation of marsh plants, where the upper limits are set by competition in relatively low-stress environments, while lower limits are set by tolerances to harsh physical conditions (Sanchez et al., 1998).

Salt marsh vegetation is not particularly well represented in New Zealand due to degradation and reclamation (Knox, 1992). Additionally, the three tidal zones have been affected differentially by disturbance and modification. Whereas the lower marsh has suffered little (except for where the introduction of *Spartina* has had an effect), the upper marsh has frequently been grazed, cleared and reclaimed (Knox, 1992).

In an attempt to gain a complete vegetational sequence for the Canterbury Region from a fragmented and modified landscape, four (somewhat fragmented) salt marshes were studied together in order to piece together an ideal "reference" salt marsh. This should be representative of the full pattern along the tidal gradient, prior to human settlement in Canterbury. The resulting template of plant distributions could then be used to guide revegetation projects and improve the chances of plant survival in Linwood Paddock restoration. Advantages of a reference wetland approach include (1) making explicit the goals of creation and restoration through identification of reference standards from data that typify sustainable conditions in the Canterbury region, (2) providing templates to which restored and created wetlands can be designed, and (3) establishing a framework whereby a decline in functions resulting from adverse impacts or a recovery of functions following restoration can be estimated (Brinson and Rheinhardt, 1996). Indeed, proper use of reference wetlands removes potential bias and provides the foundation for more objective functional-assessment procedures (Brinson and Rheinhardt, 1996).

3.2 Aims

- Determination of a continuous vegetational sequence from several fragmented systems.
- Determination of appropriate species for planting at various hydrological elevations, salinities, and pHs.
- Determination of appropriate vegetation density and substrate type expected in a mature system.
- Determination of typical levels of N, P and heavy metals, and assessment of plant tolerance ranges in the Canterbury region.
- Provision of a template by which a restored or created wetland can be designed.

3.3 Methods

Study Sites

Sampling was carried out at four widely separated salt marshes, each representing part of the fragmented ecosystem. The areas surveyed were the Heathcote Loop, the Avon River Mouth, Brooklands Lagoon and Saltwater Creek Estuary (Fig. 3.1). The location and a brief description of each is provided below. Map references are for NZMS260 M34 Amberley, M35 Christchurch and M36 Lincoln.

Heathcote Loop

Three sites were surveyed in the tidal Heathcote Loop salt marsh, now protected as Ferrymead and Settler's Reserves. These sites (map references M36 859 386, M36 862 385, and M36 865 386) are located between the Settler's Crescent light industrial area, which sits on reclaimed saltmeadow (c. 1985), and the Heathcote River itself (Fig. 3.2). The continual deposition of inorganic pollutants by the Heathcote River (e.g. plastics, bottles and packaging) (Fig. 3.8) appears to be a major threat to the wildlife. Although less species-rich, this area is close to the proposed restoration/creation site and contains almost half the total area of surviving salt marsh remnants surrounding the Avon-Heathcote Estuary. The most distinctive feature is the extensive stands consisting entirely of sea rush, *Juncus maritimus* (Fig. 3.6).

Avon River Mouth

Three sites (map references M35 877 425, M35 880 429 and M35 865 557) were surveyed on the true left bank of the Avon River, each being artificially restricted by roading or housing (Fig. 3.3). Again, household rubbish (e.g. plastic bags, bottles and paper) from river flows and domestic dumping is prevalent. This is a more species-rich area, which obviously benefits from the more sheltered location. Dominant species include: sea rush *Juncus maritimus*, jointed wire rush *Leptocarpus similis,* and three-square *Schoenoplectus pungens*. There is also an extensive bed of native musk, *Mimulus repens* (Fig. 3.9). The close proximity to the proposed site, and the high species richness despite being constrained by urbanisation and pollution, makes this a valuable reference site.

Brooklands Lagoon

Three sites (map references M34 861 713, M35 867 545 and M35 865 557) were surveyed (Fig. 3.4). These sites are relatively less modified by urbanisation however they are restricted by exotic species (e.g. *Pinus radiata*) (Fig. 3.7). The biggest impacts are from recreational users (four-wheel drive tracks and wheelings are obvious). This site is comparatively species-rich and has some of the most extensive *Schoenoplectus pungens* and *Leptocarpus similis* stands in the Canterbury region. The lagoon includes an extensive coastal barrier sand dune immediately bordering the salt marsh, low mud banks, and open brackish water. The first site has considerable freshwater influence from the Styx River.

Saltwater Creek Estuary

Four sites (map references M34 861 713, M34 865 715, M34 867 712 and M34 872 708) were surveyed (Fig. 3.5). Although somewhat windswept, this estuary is a highly productive, unpolluted area (Cromarty and Scott, 1995). It plays an important role in the support of food chains some distance from the marsh itself, and is particularly important as habitat for spawning fish (Cromarty and Scott, 1995). The estuary includes sand dunes and spits, open brackish water, a river delta, low mud banks and extensive mudflats. Saltwater Creek joins the estuary as a meandering tidal

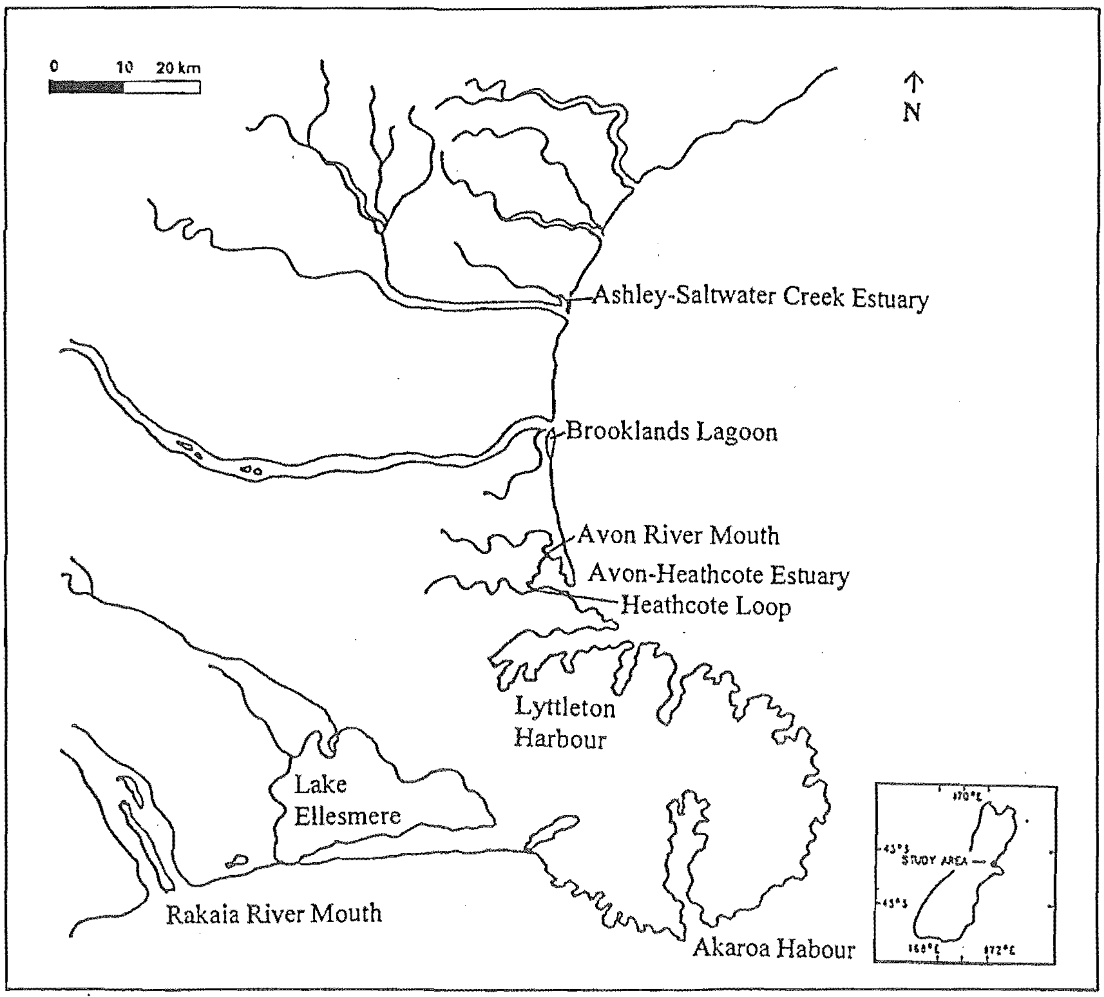
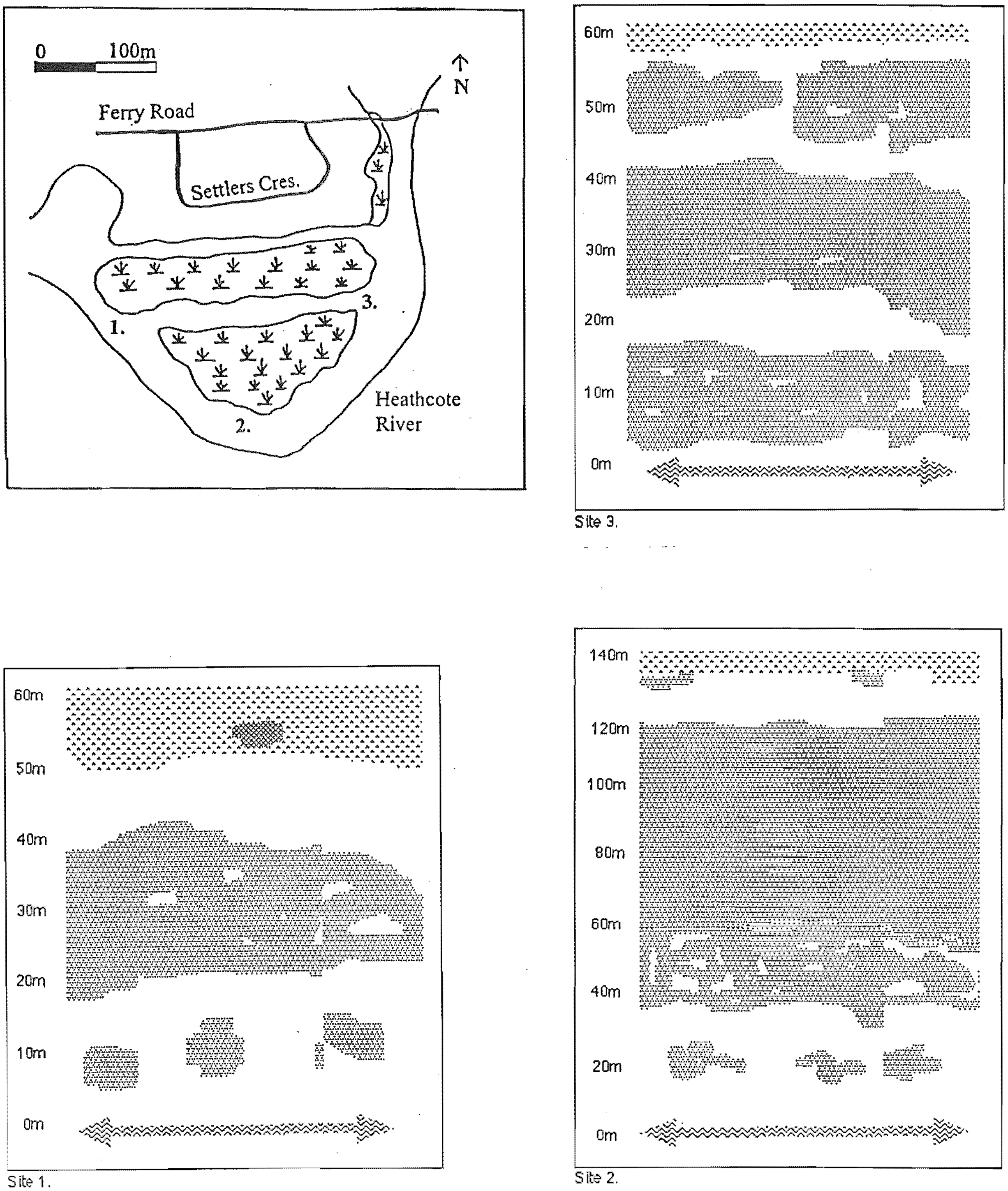


Figure 3.1. Location map of all salt marsh areas surveyed in the Canterbury region.



Key:

Low tide waterflow

Mudflat

Salt marsh

S. quinqueflora, L. dioca
A. prostratum, S. radicans

J. tenuis

J. maritimus

Figure 3.2. The salt marsh areas surveyed within the Heathcote Loop. The map numbers indicate the low tide point of each site. The distance on each site diagram is measured from the active channel at low tide.

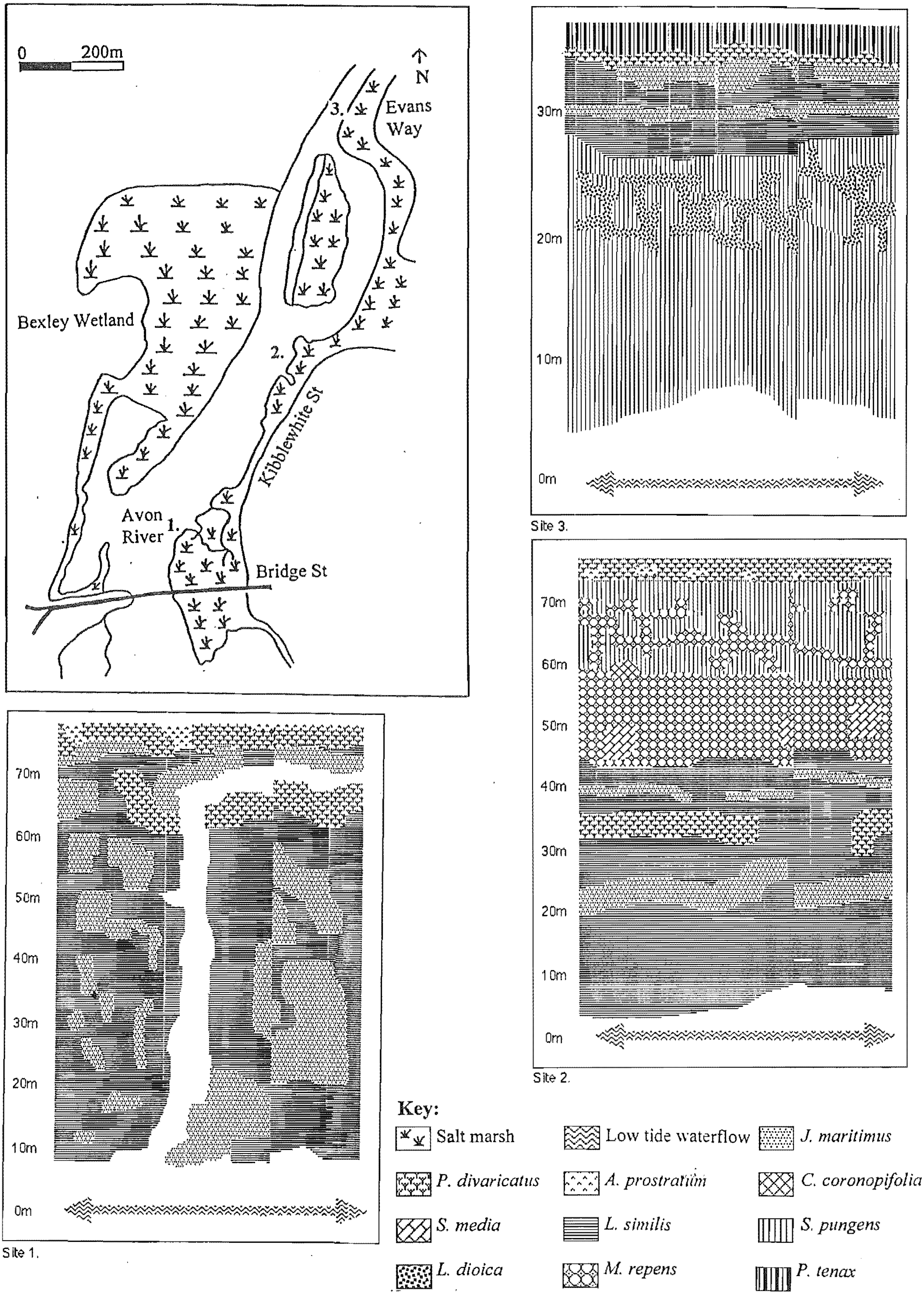


Figure 3.3. The Avon River and salt marsh areas surveyed. The map numbers indicate the low tide point at each site, from which the survey transects were run.

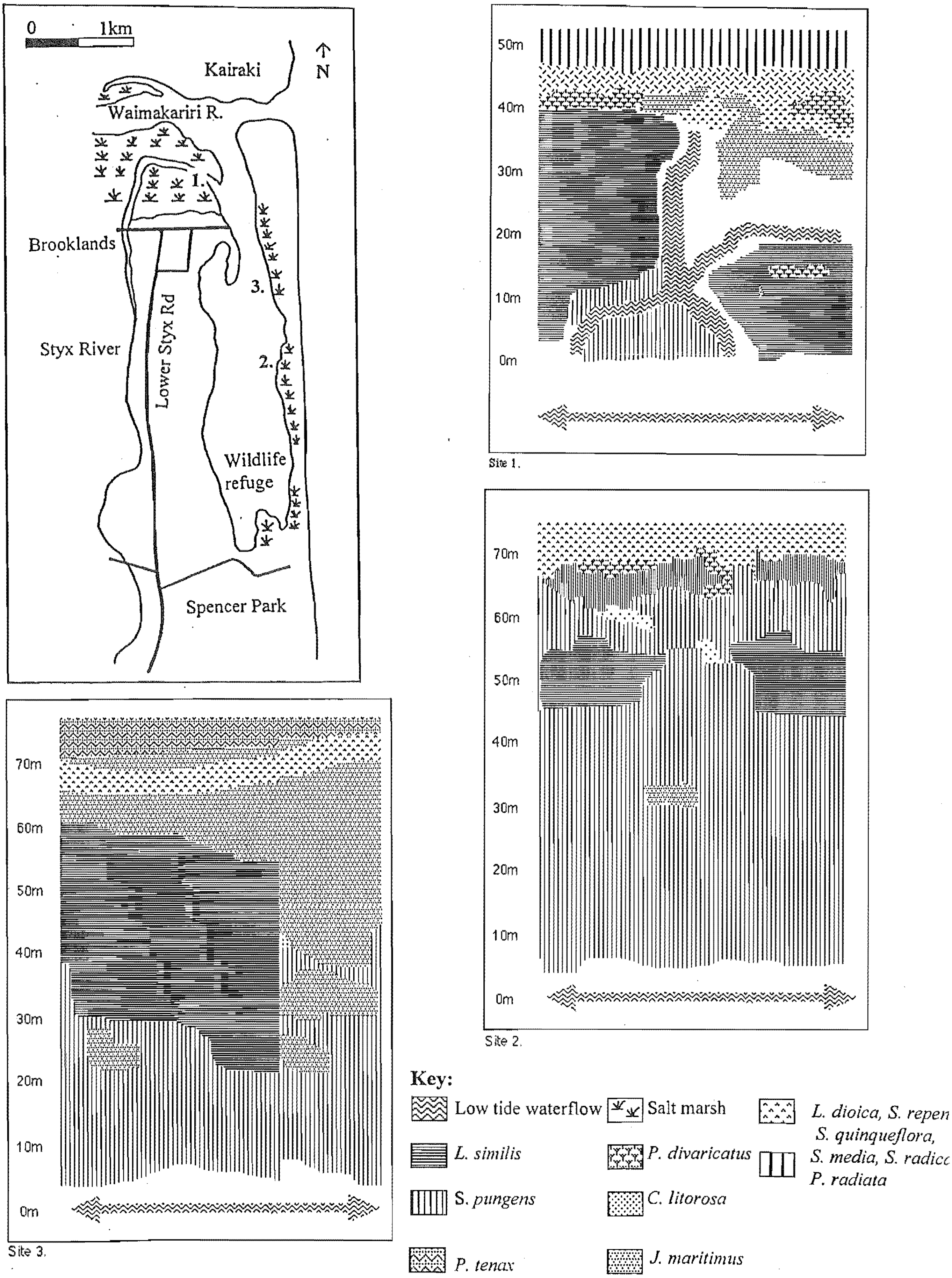


Figure 3.4. Brooklands Lagoon and Styx River salt marshes. Site numbers are located at the low tide point of each transect.

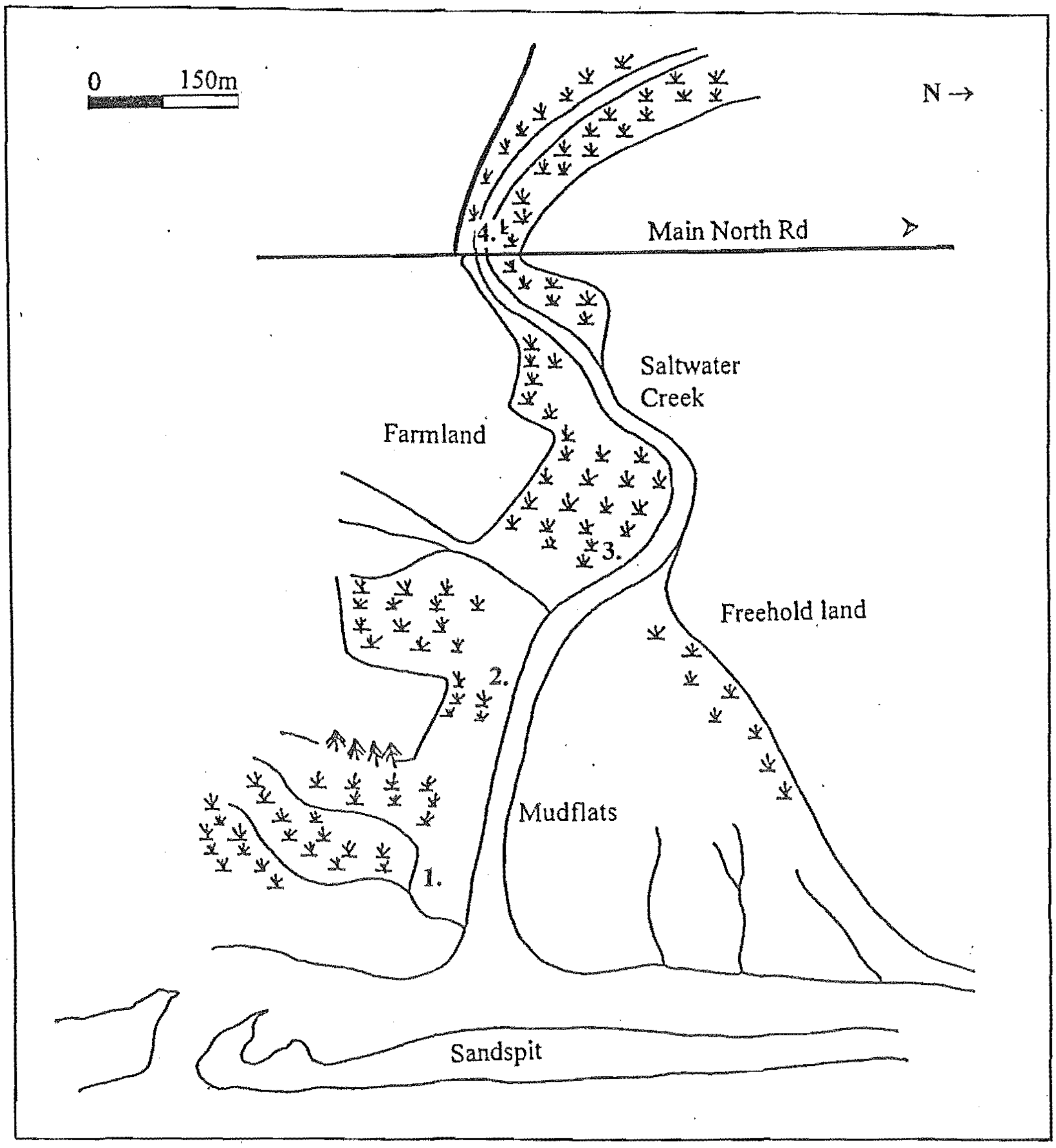
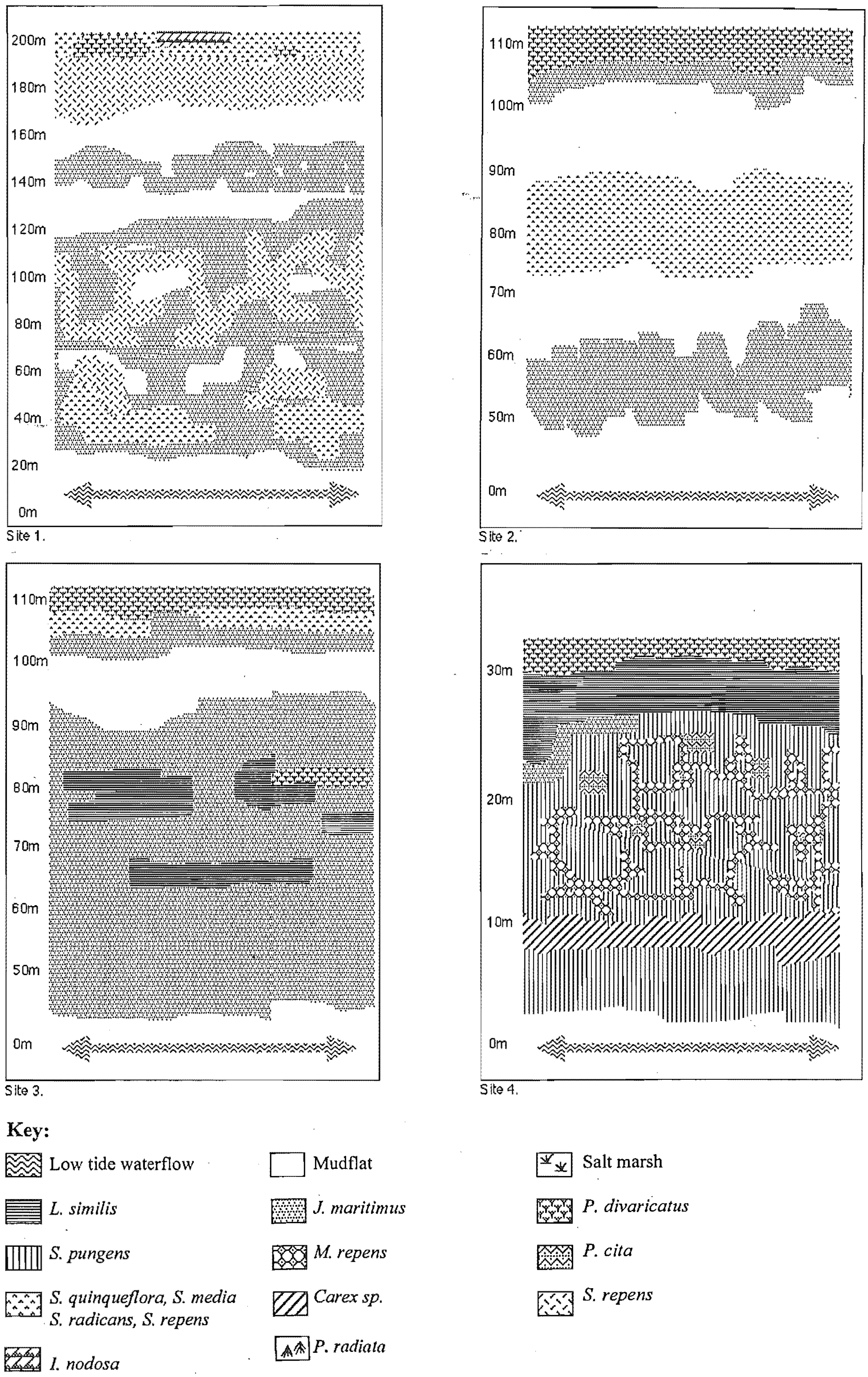


Figure 3.5. Saltwater Creek Estuary and salt marsh areas surveyed. Site numbers are located at the low tide point of each transect. The vegetation composition and cover is indicated for each site on the diagrams overleaf.


200m
180m
160m
140m
120m
100m
80m
60m
40m
20m
0m
Site 1.
110m
100m
90m
80m
70m
60m
50m
0m
Site 2.
110m
100m
90m
80m
70m
60m
50m
0m
Site 3.
30m
20m
10m
0m
Site 4.
Key:
Low tide waterflow
Mudflat
Salt marsh
L. similis
J. maritimus
P. divaricatus
S. pungens
M. repens
P. cita
S. quinqueflora, S. media
S. radicans, S. repens
Carex sp.
S. repens
P. radiata
I. nodosa

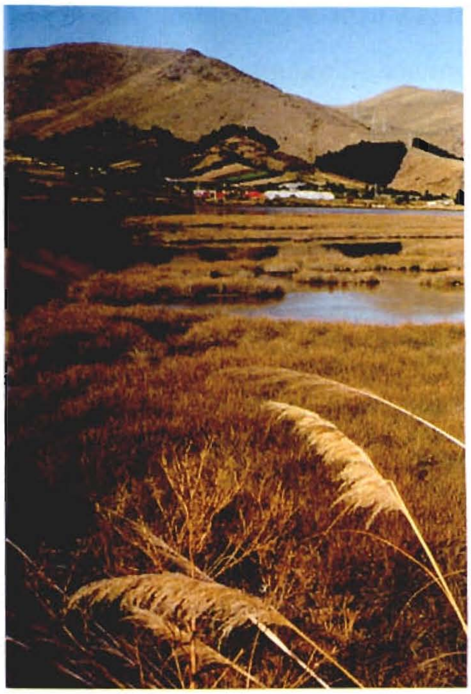


Figure 3.6. Monoculture of *Juncus maritimus* at the Heathcote sites.

Figure 3.7. *Leptocarpus similis* and *Juncus maritimus* bordered by *Pinus radiata* at Styx River, Brooklands.

Figure 3.8. Pollution and debris, at the Heathcote sites.

Figure 3.9. *Schoenoplectus pungens* (foreground), *Mimulus repens*, *Juncus maritimus* and *Leptocarpus similis* at the Avon River marshes.

swampland. Species include New Zealand flax *Phormium tenax,* sea rush *Juncus maritimus*, jointed wire rush *Leptocarpus similis,* saltmarsh ribbonwood *Plagianthus divaricatus*, three-square *Schoenoplectus pungens*, *Leptinella dioica*, and *Sarcocornia quinqueflora.*

Survey Methods

- Two surveys were conducted in marsh areas of the Avon and Heathcote Rivers, Brooklands Lagoon and Saltwater Creek Estuary. The initial survey commenced on 1 December 1997, and was followed by another beginning on 1 February 1998. Both surveys were completed within 14 days. The second survey was to enable an estimate of growth and relative success for each species in each tidal zone (low, mid and high).

- At each site in each location, two parallel transects were aligned along the gradient from high to low tide, and quadrat clusters (4 x $1m^2$) were then located at frequent intervals along the transects. At least three quadrat clusters were located along each transect (one in each tidal zone). Extra quadrat clusters were used if the vegetation changed significantly within the low, mid and high tide zones. Total number of quadrats = 320.

- The presence/absence and estimated percentage cover of each species within each quadrat was recorded. Although cover estimates are not absolute, and can be variable when performed by different people, they were used as they can serve as comparative indices for evaluating cover between different treatments and wetlands (Garbisch, 1990).

- The elevation of all sites was determined with an Abney level and corrected to the mean sea level from tide tables. Thus elevations are equivalent for all marshes.

- Soil core samples, 20 mm in diameter and 150 mm deep, were taken from each $1m^2$ quadrat and pooled from the cluster of four for analysis. Soil samples were collected in the second week of February at low tide since summer salinity was expected to be a major determinant of plant distribution. Sediment samples were analysed for salinity (ppt), pH, moisture content (%), sediment grain size (mm) and organic content (%). Analysis of N (mg/kg), P (mg/kg), and heavy metal (Cu, Cr, Ni, Zn, Pb, As, and Cd) concentration (mg/kg) was carried out under the supervision of Mike Gilson, senior scientist, Waste Management Unit, Christchurch City Council, and was determined on one pooled sample from each site.

- Soil salinity was determined on a 1:5 volume mixture of soil and distilled water. Using the method of Rhoades and Miyamoto (1990), samples were stirred for 1 min every 30 min for four hours and then covered with foil to prevent evaporation. Samples were then stored at 4°C for 24 hrs and analysed using both a conductivity meter and an ATAGO refractometer - which gave comparable results.

- Soil pH was determined on a 1:1 volume mixture of soil and distilled water. Samples were analysed two hours after mixing.

- Moisture content was determined by oven drying at 65°C for 48 hrs and expressed as a percentage of the wet weight.

- Organic content was determined from loss on ignition at 400°C for 12 hrs and expressed as a percentage of the dry weight.

- Sediment grain size was determined by dry sieving samples using 0.5, 0.125, and 0.063 mm sieves. Due to the high organic content, samples had to be treated with 4 % HCl and then oven-dried before analysis.

- All nutrients were determined on an air-dried soil sample, thus the analytical results are all expressed on a dry weight basis.

For determination of total nitrogen principle A Kjeldahl digestion was used to convert organic nitrogen to ammonium sulphate. The ammonia was removed from the digest by distillation and the concentration was determined by titration with standard acid.

For determination of total phosphorus the sample was treated with concentrated nitric acid and concentrated perchloric acid in order to release the organic phosphorus (S208 digestion). An acid ammonium molybdate reagent was added to a suitable volume of the filtered, digested sample. The resulting phosphomolybdate complex was reduced to an intense blue colouration by ascorbic acid and the intensity of the colour was spectrophotometrically determined at 882 nm and converted to phosphorus concentration against a standard curve.

- All metal analysis was determined on a dry weight basis. Samples (1-2 g) were prepared using 4 ml concentrated nitric acid + 2.5 ml concentrated perchloric acid (60 %) and made up with 20 ml distilled water. Digestion was carried out at 90°C on a calibrated hot plate for 2 hrs. The resulting sample was then filtered through a 9cm GA-100 glass fibre paper.

For determination of Cu, Cr, Ni, Zn, and Pb concentration, atomic absorption spectroscopy was used.

For determination of Cd and As concentration, graphite furnance atomic absorption spectroscopy was used.

3.4 Results

Vegetation Versus Elevation (individual species)

The major native salt marsh species surveyed in Canterbury are distributed from 1.68 to 2.65 m above mean sea level (Fig. 3.10). The elevation range of species occurring at more than one location, differs between locations (Table 3.1). Although most species distributions overlap between locations, few locations appear to enable the entire species range indicated by the other locations (Table 3.1). Fig. 3.10 shows the continuous distributions of some of the major species according to elevation. These distributions have been compiled from the fragmented distributions recorded at each location. *L. similis* and *P. divaricatus* have the greatest tidal elevation range. Salt marsh dominants *J. maritimus*, and *S. pungens* also have extensive ranges. Mid tidal elevations are characterised by the presence of *L. dioica, P. cita, M. repens* and *C. coronopifolia* (Table 3.1). There is an increase in the number of species (e.g. *C. litorosa, S. repens, S. quinqueflora, S. radicans* and *P. tenax*) and cover, approximately 2.15 m above mean sea level (MSL) (Fig. 3.10). For the majority of species (except *J. maritimus* and *L. similis*), maximum biomass and probability of occurrence are located near the upper end of each species' elevation range (Fig. 3.10).

Vegetation Versus Salinity (individual species)

The salinity range of species occurring at more than one location, differs between locations (Table 3.2). Although most species distributions overlap between locations, few locations appear to enable the entire species salinity range indicated by the other locations (Table 3.2). Fig. 3.11 shows the continuous distributions of some of the major species according to salinity. These distributions have been compiled from the fragmented distributions recorded at each location. The distribution of the major species according to salinity shows that none extends its range beyond 42 ppt (Fig. 3.11). *S. radicans, S. repens* and *S. quinqueflora* exhibit the greatest salinity ranges. These three species were observed at salinities 15 ppt higher than the upper limit recorded for most other species (Fig. 3.11). *J. maritimus, L. similis* and *S. pungens* also have extensive salinity ranges, but these are concentrated towards the lower limits of the observed salinity range for all species surveyed. The remaining species

Table 3.1. Elevation range (m above MSL) of native salt marsh species at all locations.

Location	Heathcote River	Avon River	Brooklands Lagoon	Saltwater Creek
Juncus maritimus	1.96 - 2.28	1.98 - 2.15	1.91 - 2.35	1.89 - 2.65
Leptocarpus similis	-	1.88 - 2.27	1.68 - 2.35	2.13 - 2.65
Mimulus repens	-	1.88 - 2.16	-	1.87 - 1.90
Plagianthus divaricatus	2.28	2.15 - 2.27	1.68 - 2.15	2.16 - 2.65
Spergularia media	-	1.88 - 2.16	2.15	2.10 - 2.37
Schoenoplectus pungens	-	1.85 - 2.14	1.68 - 2.35	1.80 - 2.16
Sarcocornia quinqueflora	2.16 - 2.35	-	2.15	2.30 - 2.37
Selliera radicans	2.28 - 2.35	-	2.15 - 2.35	1.89 - 2.35
Samolus repens	2.35	-	2.15 - 2.17	2.10 - 2.37
Poa cita	-	-	-	1.89 - 1.91
Cotula coronopifolia	-	1.88 - 2.16	-	-
Leptinella dioica	-	1.88 - 2.16	-	-
Carex litorosa	-	-	2.19 -2.35	-
Phormium tenax	-	-	2.35	-

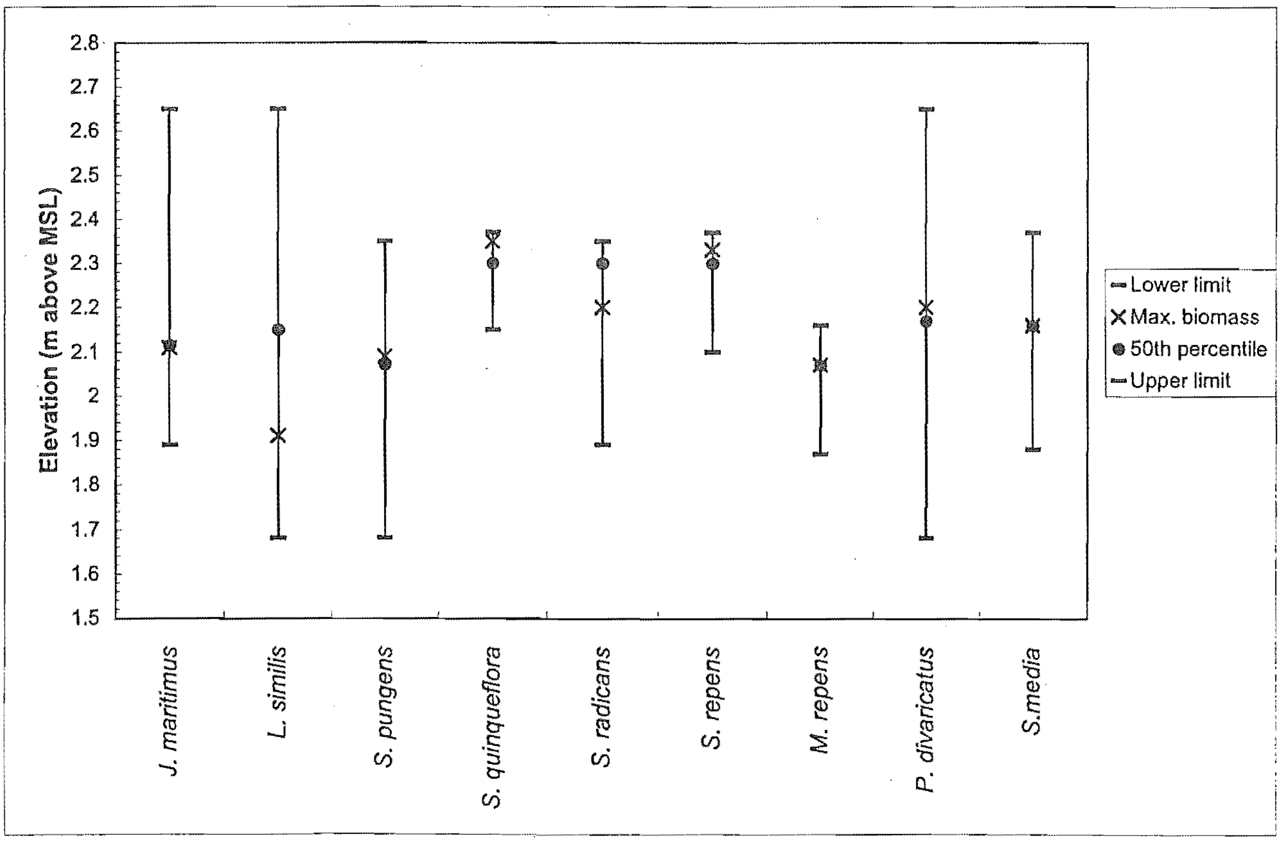


Figure 3.10. Elevation range and location of maximum biomass of native salt marsh species (compiled from all sites).

Table 3.2. Salinity range (ppt) of native salt marsh species at all locations.

Location	Heathcote River	Avon River	Brooklands Lagoon	Saltwater Creek
Juncus maritimus	9 - 30	12 - 24	6 - 24	13.5 – 30
Leptocarpus similis	-	12 - 24	12 - 18	15 – 21
Mimulus repens	-	18 - 24	-	18 – 24
Plagianthus divaricatus	30	12 - 24	18	13.5 – 24
Spergularia media	-	18 - 21	18	13.5 – 30
Schoenoplectus pungens	-	18 - 30	6 - 24	12 – 24
Sarcocornia quinqueflora	24 - 42	-	18	13.5 – 30
Selliera radicans	30 - 42	-	18 - 24	24
Samolus repens	42	-	12 - 18	18 – 30
Poa cita	-	-	-	18 – 24
Cotula coronopifolia	-	18 - 24	-	-
Leptinella dioica	-	12 - 30	-	-
Carex litorosa	-	-	12 - 30	-
Phormium tenax	-	-	12 - 30	-

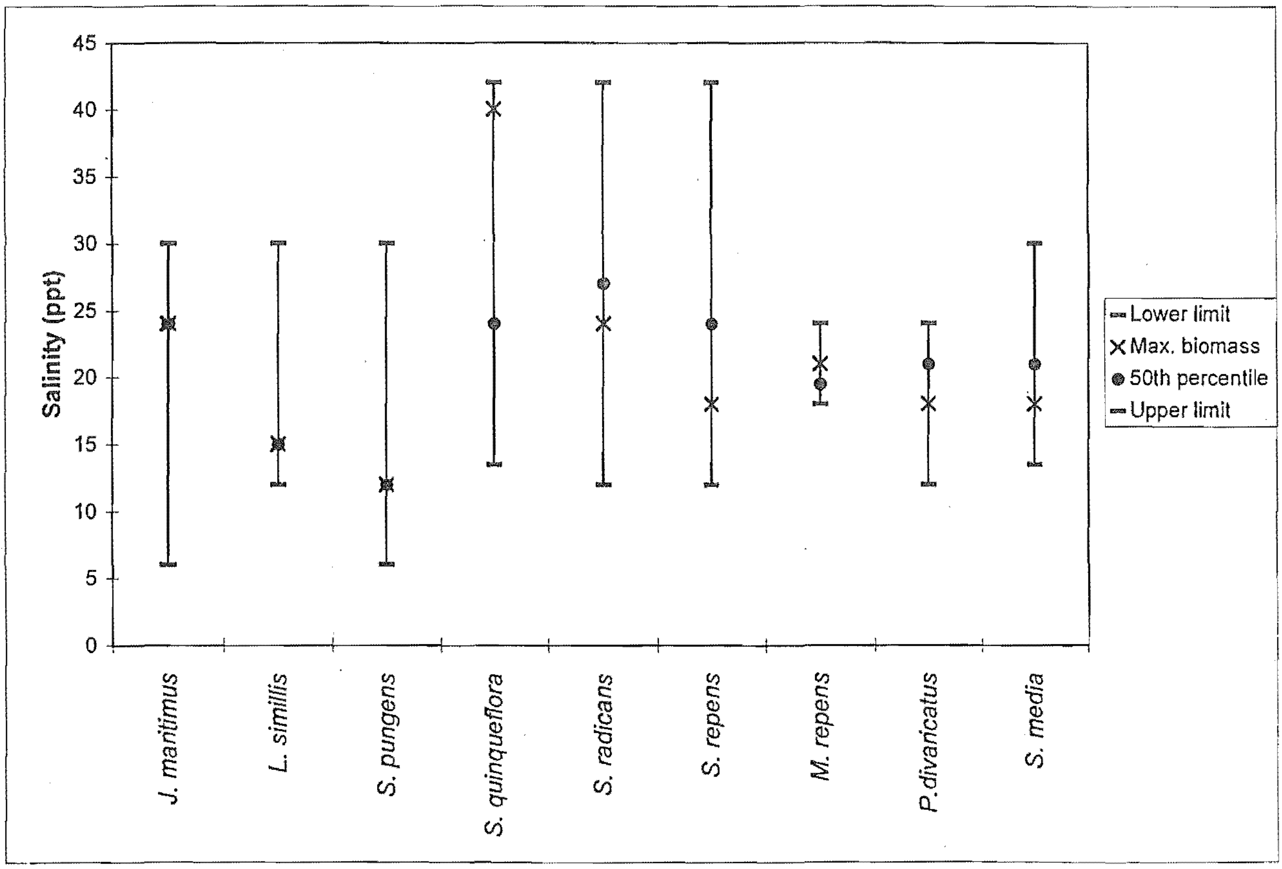


Figure 3.11. Salinity range and location of maximum biomass of native salt marsh species (compiled from all sites).

are observed at salinities within the range of 12 – 30 ppt. Except for *S. quinqueflora*, the location of all species maximum biomass is recorded at the 50^{th} percentile or in the lower half of each species salinity distribution.

Vegetation Versus pH (individual species)

The distribution of the major salt marsh species with respect to pH is within the range of 5.3 - 8.1 (Fig. 3.12). *J. maritimus* is the only species distributed over the entire range. *M. repens*, *L. similis*, *S. pungens* and *S. quinqueflora* have a pH range of approximately 2, but all other species have a very narrow pH range tending towards neutrality or slightly acidic (Fig. 3.12).

Vegetation Cover Estimates

Percent cover of each species varies significantly within sites and between locations (Tables 3.3 - 3.6). The between-site, within-location, variation is less than the between-location variation. Furthermore, the relative percent cover of each species at each location, in relation to the other species present, is consistent between sites and locations. Table 3.7 shows the mean percent cover of each species in each tidal zone. This indicates the relative proportions expected of each species at each zone in a mature marsh system. *L. similis* has the highest percent cover of any species in the low and high tide zones, with percent cover approaching 100 %. *S. pungens* is the only other species to record its highest percent cover in the low tide zone (Table 3.7). *J. maritimus*, *M. repens*, *S. media*, and *S. quinqueflora* all show the highest percent cover in the mid tide zone. All remaining species, however, recorded the highest percent cover in the high tide zone (Table 3.7). There was no significant difference in percent cover over the growing season for any species as measured by a paired sample t-test ($P > 0.05$).

Sediment Analysis

The majority of salt marsh substrate was composed of the silt/clay fraction, followed by fine-medium sand (Table 3.8). The organic component is variable both within and between sites, with some soils classed as organic (> 5 % organic matter), and some as mineral (< 5 % organic matter). Except for Avon River sites, the highest zone at each location had the highest organic matter content (Table 3.8). At most sites the

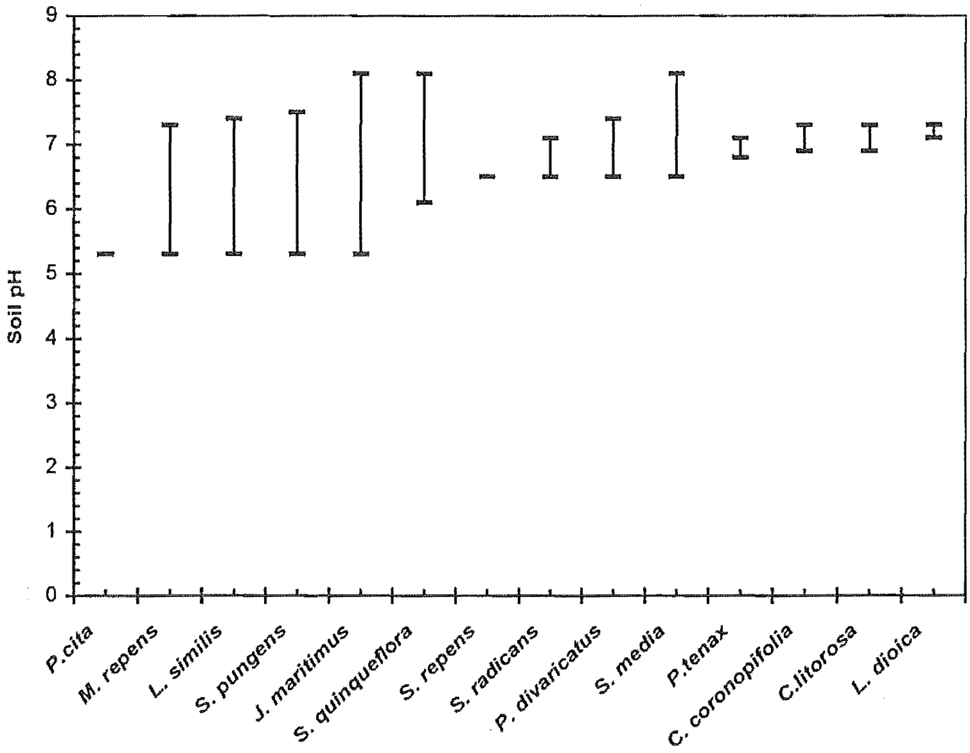


Figure 3.12. pH range of native salt marsh species, compiled from all sites.

Table 3.3. Percentage species cover from quadrats samples at the Heathcote River mouth, December 1997 and February 1998.

Site	1			2			3		
Zone	Low	Mid	High	Low	Mid	High	Low	Mid	High
Species cover	%	%	%	%	%	%	%	%	%
J. maritimus	36.25	51.88	-	34.88	83.38	40.00	89.75	62.75	-
S. quinqueflora	-	-	68.00	-	-	3.00	-	-	43.33
S. radicans	-	-	22.50	-	-	14.63	-	-	-
S. repens	-	-	4.50	-	-	-	-	-	-

Table 3.4. Percentage species cover quadrats sampled from the Avon River, December 1997 and February 1998.

Site	1			2			3		
Zone	Low	Mid	High	Low	Mid	High	Low	Mid	High
Species cover	%	%	%	%	%	%	%	%	%
J. maritimus	5.25	2.00	27.00	2.00	-	83.00	-	-	97.50
L. similis	98.0	71.88	55.0	99.75	25.00	-	-	100.0	48.25
P. divaricatus	10.00	31.86	34.38	-	-	-	-	-	51.50
C. coronopifolia		-	-	-	-	9.00	1.00	-	-
L. dioica	-	-	-	-	-	-	-	-	10.00
S. media	-	-	-	-	33.00	-	6.33	-	-
M. repens	-	-	-	-	90.13	45.75	16.25	-	-
S.pungens	-	-	-	-	-	35.66	50.94	46.25	-

Table 3.5. Percentage species cover from quadrats sampled at Brooklands Lagoon, December 1997 and February 1998.

Site	1			2			3		
Zone	Low	Mid	High	Low	Mid	High	Low	Mid	High
Species cover	%	%	%	%	%	%	%	%	%
J. maritimus	-	5.00	10.25	-	53.00	6.00	-	15.00	-
L. similis	93.75	78.50	49.25	-	-	76.63	-	-	95.63
S. pungens	46.75	54.25	-	100.0	41.50	24.63	100.0	53.13	40.71
P. divaricatus	25.00	-	46.00	-	-	-	-	-	-
S. radicans	-	-	79.00	-	-	72.50	-	-	2.00
S. repens	-	-	-	-	-	9.00	-	-	-
S. quinqueflora	-	-	22.50	-	-	-	-	-	-
S. media	-	-	3.00	-	-	-	-	-	-
C. litorosa	-	-	-	-	-	30.00	-	23.50	-
P. tenax	-	-	-	-	-	-	-	-	7.00

Table 3.6. Percentage species cover from quadrats sampled at Saltwater Creek Estuary, December 1997 and February 1998. Sites are numbered from Main North Rd.

Site	**1**			**2**			**3**			**4**		
Zone	Low	Mid	High	Low	Mid	High	Low	Mid	High	Low	Mid	High
Species cover	%	%	%	%	%	%	%	%	%	%	%	%
J. maritimus	12.66	29.0	1.00	47.88	-	12.00	66.13	53.00	30.25	-	-	-
L. similis	-	-	-	-	-	-	-	59.33	-	-	24.75	75.20
S. pungens	-	-	-	-	-	-	-	-	-	42.81	20.50	9.00
P.divaricatus	-	-	-	-	-	47.66	-	18.00	7.50	-	-	-
S. radicans	-	-	12.50	-	4.33	-	-	-	2.00	-	-	-
S. repens	35.75	53.25	32.78	-	16.80	-	-	-	-	-	-	-
S. quinqueflora	19.50	-	9.75	-	29.50	7.5	-	-	-	-	-	-
P. cita	-	-	-	-	-	-	-	-	-	-	3.30	34.00
I. nodosa	-	-	6.00	-	-	-	-	-	-	-	-	-
M. repens	-	-	-	-	-	-	-	-	-	-	51.25	35.25
S. media	7.00	6.00	-	-	4.2	-	-	-	-	-	-	-

Table 3.7. Mean percent cover of native salt marsh species in each zone across all locations and sites.

Zone	**Low**	**Mid**	**High**
Species cover	%	%	%
Juncus maritimus	36.85	39.45	30.90
Leptocarpus similis	97.17	59.00	91.00
Sarcocornia quinqueflora	19.50	29.50	25.68
Selliera radicans	-	4.33	33.60
Samolus repens	-	-	6.75
Plagianthus divaricatus	17.50	24.93	34.92
Cotula coronopifolia	-	1.00	6.75
Leptinella dioica	-	-	10.00
Spergularia media	6.67	19.50	3.00
Mimulus repens	16.25	70.69	40.50
Schoenoplectus pungens	46.83	43.13	24.78
Phormium tenax	-	-	7.00
Carex litorosa	-	23.50	30.00
Poa cita	-	3.30	34.00

Table 3.8. Mean percentage particle size distribution of each zone at each location.

Location	**Heathcote River**			**Avon River**			**Brooklands Lagoon**			**Saltwater Creek**		
Zone	Low	Mid	High	Low	Mid	High	Low	Mid	High	Low	Mid	High
Particle Size	%	%	%	%	%	%	%	%	%	%	%	%
> 0.500 mm Coarse sand	1.67	0.63	8.93	1.03	1.70	0.87	0.51	0.39	0.29	1.79	7.43	1.82
0.500 - 0.125 mm Fine – medium sand	4.86	5.38	6.40	26.69	39.91	39.81	16.77	23.03	21.35	19.00	13.36	29.24
0.125 - 0.063 mm Very fine sand	4.00	5.18	4.99	3.34	2.60	3.14	0.79	0.94	2.88	17.91	10.82	11.16
< 0.063 mm Silt/clay	85.13	84.5	69.48	60.40	48.80	49.18	78.54	70.05	59.81	61.64	62.43	50.97
Organic component	5.35	4.32	10.21	9.83	7.00	6.98	3.39	5.74	15.67	4.15	5.97	6.81

maximum proportion of silt/clay sediment is in the lowest zone, with the higher zones containing a higher proportion of the coarser sediment (Table 3.8).

Heavy Metal and Nutrient Analysis

Mean levels for nitrogen, phosphorus and all heavy metals except nickel were highest at the Heathcote River sites and lowest at the Saltwater Creek sites (Table 3.9). Indeed, heavy metal and nutrient levels show a general decline with increasing distance from the Christchurch urban centre (Table 3.9). This decline is most obvious for chromium, zinc and lead. The between-site, within-location variation for all metals except nickel, and for both nutrients, is less than that observed between locations.

Species' Tolerance to Heavy Metal Levels

Individual species tolerance to heavy metals was inferred from levels measured where each species occurred at each location (Table 3.10). Though the heavy metal range each species can tolerate may be far greater, it would best be determined by experimentation and is outside the scope of this study. Table 3.10 does, however, give the heavy metal range salt marsh species would normally grow under in Canterbury.

Species' Optimum Distribution Template

Fig. 3.13 shows the optimum distribution of salt marsh species with respect to salinity and elevation, determined by the location of maximum biomass with a threshold of $p > 0.5$. This provides a template from which restored and created wetlands can be designed in the Canterbury region. There is overlap between the optimal location for all species except *L. similis* and *S. quinqueflora.*

Table 3.9. Heavy metal and nutrient level (mg/kg) determined on one sample pooled from each site.

Site	*Cu* *mg/kg*	*Cr* *mg/kg*	*Ni* *mg/kg*	*Zn* *mg/kg*	*Pb* *mg/kg*	*As* *mg/kg*	*Cd* *mg/kg*	*TKN* *mg/kg*	*TOP* *mg/kg*
Heathcote 1	7.76	14.51	8.79	63.8	22.48	6.86	0.03	975	564
Heathcote 2	17.95	37.36	13.41	133.0	74.06	4.95	0.02	2167	1113
Heathcote 3	13.17	30.20	11.35	102.0	41.95	4.26	0.01	1668	974
Avon 1	26.87	33.85	14.97	138.0	44.72	3.63	0.07	2590	815
Avon 2	27.60	32.50	13.9	121.0	59.69	5.93	0.03	2979	1262
Avon 3	8.40	11.80	7.8	68.4	19.8	2.26	0.02	799	513
Brooklands 1	9.87	11.31	12.75	60.0	15.01	3.25	0.02	748	622
Brooklands 2	5.02	9.24	9.44	40.0	10.25	3.01	0.01	1227	551
Brooklands 3	11.73	11.73	14.62	73.5	20.38	4.94	0.01	2728	792
Saltwater Creek 1	6.72	8.55	9.57	40.1	9.16	2.34	0.01	544	391
Saltwater Creek 2	6.37	8.36	9.16	39.0	8.96	2.47	0.01	660	471
Saltwater Creek 3	9.34	10.17	11.42	57.7	18.48	3.36	0.01	2084	826
Saltwater Creek 4	11.35	12.17	14.03	71.2	16.92	3.65	0.02	2181	812

Table 3.10. Metal tolerance (mg/kg) of native salt marsh species at all locations.

Location	Cu	Cr	Ni	Zn	Pb	As	Cd
Juncus maritimus	5.02 - 27.60	8.36 - 37.36	7.80 - 14.97	39.0 - 138.0	8.96 - 74.06	2.26 - 6.86	0.01 - 0.07
Leptocarpus similis	5.02 - 27.60	8.36 - 33.85	7.80 - 14.97	39.0 - 138.0	8.96 - 59.69	2.26 - 6.86	0.01 - 0.07
Mimulus repens	6.37 - 27.60	8.36 - 33.85	7.80 - 14.97	39.0 - 138.0	8.96 - 59.69	2.26 - 6.86	0.01 - 0.07
Plagianthus divaricatus	5.02 - 27.60	8.36 - 37.36	7.80 - 14.97	39.0 - 138.0	8.96 - 74.06	2.26 - 6.86	0.01 - 0.07
Spergularia media	5.02 - 27.60	8.36 - 33.85	7.80 - 14.97	40.0 - 138.0	8.96 - 59.69	2.26 - 6.86	0.01 - 0.07
Schoenoplectus pungens	5.02 - 27.60	8.36 - 37.36	7.80 - 14.97	40.0 - 138.0	8.96 - 59.69	2.26 - 6.86	0.01 - 0.07
Sarcocornia quinqueflora	6.37 - 17.95	8.36 - 37.36	8.79 - 14.97	39.0 - 138.0	8.96 - 74.06	2.34 - 6.86	0.01 - 0.03
Selliera radicans	6.37 - 17.95	8.36 - 37.36	8.79 - 14.97	39.0 - 138.0	8.96 - 74.06	2.34 - 6.86	0.01 - 0.03
Samolus repens	6.37 - 17.95	8.36 - 37.36	8.79 - 14.97	39.0 - 138.0	8.96 - 74.06	2.34 - 6.86	0.01 - 0.03
Poa cita	6.37 - 11.35	8.36 - 12.17	9.16 - 14.03	39.0 - 71.2	8.96 - 16.92	2.34 - 3.65	0.01 - 0.02
Cotula coronopifolia	8.40 - 27.60	11.80 - 33.85	7.80 - 14.97	68.4 - 138.0	19.8 - 44.72	2.26 - 5.93	0.02 - 0.07
Leptinella dioica	8.40 - 27.60	11.80 - 33.85	7.80 - 14.97	68.4 - 138.0	19.8 - 44.72	2.26 - 5.93	0.02 - 0.07
Carex litorosa	5.02 - 11.73	9.24 - 11.73	9.42 - 14.62	60.0 - 73.5	10.25 - 20.38	3.01 - 4.94	0.01 - 0.02
Phormium tenax	5.02 - 11.73	9.24 - 11.73	9.42 - 14.62	60.0 - 73.5	10.25 - 20.38	3.01 - 4.94	0.01 - 0.02
Observed heavy metal range	5.02 - 27.60	8.36 - 37.36	7.80 - 14.97	39.0 - 138.0	8.96 - 74.06	2.26 - 6.86	0.01 - 0.07

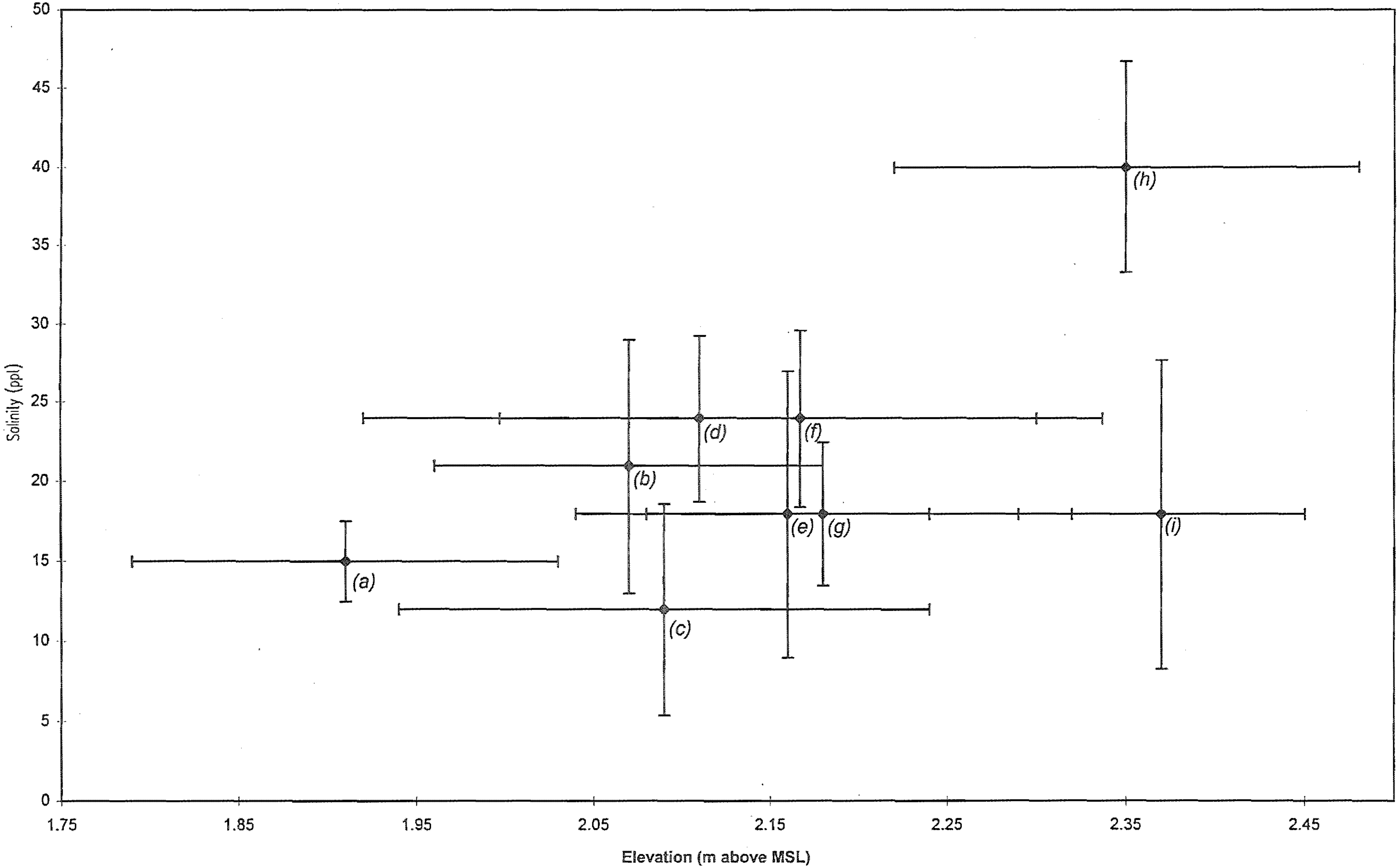


Figure 3.13. Template of species optimal distribution with respect to elevation (m above MSL) and salinity (ppt). Optimal distribution is determined by maximum biomass with a threshold of $p > 0.5$. (a) *L. similis*, (b) *M. repens*, (c) *S. pungens*, (d) *J. maritimus*, (e) *S. media*, (f) *S. radicans*, (g) *P. divaricatus*, (h) *S. quinqueflora*, (i) *S. repens*.

3.5 Discussion

Vegetation Versus Elevation

The lower marsh in the Canterbury region is characterised primarily by *Leptocarpus similis*. *Juncus martimus* or *Schoenoplectus pungens* may dominate this zone if *Leptocarpus similis* is absent from a site. These species usually exist as a monoculture, although sometimes *Juncus maritimus* and *Leptocarpus similis* will coexist with *Leptocarpus similis* being the dominant. The middle marsh is characterised by the addition of *Poa cita, Cotula coronopifolia* and *Mimulus repens*. The dry upper marsh is characterised by numerous salt-tolerant herbaceous species including *Sarcocornia quinqueflora*. The upper marsh fringe typically consists of *Plagianthus divaricatus* and *Phormium tenax*.

For the majority of species, maximum biomass and probability of occurrence is located towards the upper end of each species' elevation range (Fig. 3.10) i.e. the realised niche is asymmetrical compared to the fundamental niche. Indeed, mean high tide is generally accepted as an important point separating the middle and upper marsh zones and is characterised by a sudden increase in the number of species and cover (Chapman, 1974). Such an increase was recorded in this study at ~2.15 m above mean sea level. Mean high tide for the Canterbury region is ~2.2 m above mean sea level (The Press, tide forecasts); therefore in this study, plant distributions are distributed at slightly lower elevations than other studies have shown. In addition, the distribution of *Leptocarpus similis* and *Schoenoplectus pungens* is approximately 0.5 m lower than that shown by Partridge and Wilson (1989) for these species in Otago salt marshes. However, the zonation of the remaining salt marsh species surveyed in Canterbury marshes is analogous to that found by Partridge and Wilson (1989) for Otago salt marshes.

Vegetation Versus Salinity

Salinity varies non-linearly throughout the marsh. Lower marsh soils are frequently flooded and have a fairly constant salinity approximating that of the flooding sea water. Conversely the upper marsh, or raised sections may only be occasionally flooded, and depending on the rainfall and freshwater inflow, may have elevated or depressed salinities.

All salt marsh species in this study were distributed within salinities ranging from 6 – 42 ppt. Typical salt marsh salinities range from 33 - 42 ppt on a daily basis, to less than 30 ppt and more than 50 ppt on an annual basis (Phleger, 1977; Pomeroy and Wiegert, 1981; Long and Mason, 1983). *S. quinqueflora* was the only species to record maximum biomass at high salinities, nearing the limit of its observed tolerance. For most species, maximum biomass was located toward the lower salinities of each species' range. Again, the realised niche is asymmetrical to the fundamental niche. Generally excess salinity dwarfs plants, such reduction being ascribed to the high osmotic pressures engendered in the cells (Chapman, 1976). There are, however, other possibilities, such as water-logging of very clayey soils through the effect of sodium on soil colloid dispersion, with the result that reducing conditions inimical to the plants are produced (Chapman, 1976). Further possibilities involve ion antagonism and the effect of sodium on the calcium metabolism (Chapman, 1976).

Some sites had considerable riverine input and recorded salinities lower than generally recorded in a salt marsh (e.g. 6 ppt). However, lack of salinity is not as much of a concern as hypersalinity. Partridge and Wilson (1987) determined that some salt marsh species have a salt requirement for maximum growth, but that most grow best in freshwater. No species requires saline solutions to survive. Salinity is highest in summer due to heightened evapotranspiration and low rainfall (Clarke and Hannon, 1969). Therefore salt marsh plants grow during the season of highest salinity. As Fig. 3.11 shows, salt marsh species vary in their tolerance to salinity and are intolerant of extreme salinities. Therefore summer salinities are a major determinant of salt marsh zonation. Indeed, Partridge and Wilson (1989), found that the gradients in the vegetation best matched the salt-tolerance of the species, and species of like tolerance were grouped together in the marsh.

The effect of salinity is also dependent on the age of each individual plant. Seedlings and annual plants are more likely to be affected than mature plants.

Vegetation Versus pH

The salt marsh species surveyed were distributed over the entire pH range recorded in salt marsh sediments (Fig. 3.12). However this was a very narrow range (5.3 – 8.1) and may not reflect individual species tolerances. It is widely accepted that the pH of wetland soils tends towards neutral (pH 7) (Pomeroy and Wiegert, 1987), and that the daily and yearly range is similar, varying from approximately 6.8 to 8.3 (Long and Mason, 1983).

Vegetation Cover

Vegetation cover estimates (Tables 3.3 - 3.6) proved to be no better an indicator than simple presence/absence data in such mature marsh systems. As these are clonal species which spread mostly vegetatively (if seed is dispersed, it is the tide which carries it), their distribution is "blocky" consisting of monocultural clonal stands at the scale sampled (1 m^2) rather than mixed-species patches. Therefore if a species is present in a mature marsh, cover is likely to be approaching 100 % (Figs. 3.2 - 3.5).

Salt Marsh Sediments

Salt marsh sediments surveyed in the Canterbury marshes were predominantly composed of the silt/clay fraction (Table 3.8). Certainly, grain size in marshes depends upon what sizes are available. However, marshes do accumulate the finest sediment available in the system due to the low current velocities in the very shallow water (Phleger, 1977). Deposition of fine-grained sediment is also aided by the dense vegetation in marshes which acts as a baffle to further reduce water velocities (Phleger, 1977).

The majority of salt marsh sediments surveyed would be classed as organic, especially in the high tide zone (Table 3.8). This trend is consistent with that found for most other salt marshes, where organic content is sourced from both the plants *in situ* and also the tidal wash. Indeed, Long and Mason (1983) state that although variation exists at all levels, salt marsh organic content generally increases with increasing marsh elevation.

Heavy Metal and Nutrient Level

Saltwater Creek and Brooklands Lagoon, located furthest from the Christchurch City centre or any other urban influence, recorded the lowest levels of both heavy metals and nutrients (Table 3.9). Generally, anaerobic salt marsh soils have been shown to act as sinks for heavy metals, holding significant quantities of Zn, Fe, and Mn

(Elderfield and Hepworth, 1975). Indeed, all levels recorded exceed normal ranges found for inland soils by Bidwell (1979). However, all levels are within the range of those determined for unpolluted marshes by Kay and Rajanvipart (1977) i.e. using these limits, all marshes surveyed would be classified as unpolluted. Such levels were determined from British marshes, and may not be relevant to the New Zealand situation. The difference in heavy metal and nutrient level between the urban Avon and Heathcote marshes, and the more removed Brooklands Lagoon and Saltwater Creek marshes (Table 3.9), suggests that the former sites are polluted in this context.

The availability of nutrients and excess nutrients also affect plant productivity and zonation. Seawater is high in Mg, Ca, K, and S, providing adequate amounts of these nutrients to salt marsh vegetation (Broome, 1990). Phosphorus is an important mineral, although it is not considered limiting in salt marshes due to its relative abundance and biochemical stability under neutral pH conditions (Mitsch and Gooselink, 1993; Pomeroy and Wiegert, 1981). Nitrogen levels are somewhat variable, and high levels may effect zonation resulting in lowered species diversity.

Levine et al. (1998) examined the effects of nutrient availability on the competitive interactions of New England salt marsh perennials that occupy discrete vegetational zones parallel to the shoreline. They found that under conditions of nutrient limitation, competitive dominance results from efficient competition for nutrients. This may cause a nutrient-induced reversal in the competitive dynamics among salt marsh perennials and result in modification of plant zonation in marshes. Typically, nitrogen fertilisation increased the absolute biomass of the ambient marsh inferior, while decreasing the biomass of the dominant (Levine et al., 1998). Thus, the competitive inferior under ambient conditions dominated plots under nutrient-enhanced conditions (Levine et al., 1998). The most dramatic prediction from these results is that under enhanced nutrient conditions, where a species is both the best competitor and the most tolerant of physical stresses, the entire salt marsh could become dominated by this one species (Levine et al., 1998). Without nutrient limitation, the striking zonation of salt marshes may breakdown. This prediction has potential implications for conservation and management, particularly in marshes subject to high nutrient levels from anthropogenic sources. Although nutrient tests were necessarily limited in number, high nutrient levels did not appear to have altered native salt marsh species zonation in the marshes surveyed. However, this relationship may be evident if one compares introduced or exotic species with natives. For example at Saltwater Creek there are developed areas where pasture species dominate salt marsh species, adjacent to the estuary. In such areas the distribution of salt marsh species is limited to the lower elevations while pasture species dominate the less physically stressful, fertilised, higher elevations.

Species' Optimum Distribution Template for Revegetation

The species' optimum distribution template indicates which species to plant at various elevation and salinity levels in a created or restored marsh. Since the distributions of *L. similis* and *S. quinqueflora* do not overlap with that of any other species, they would be expected to grow in extensive monocultures. *L. similis* would be expected to dominate the low elevation range, and *S. quinqueflora* would be expected to dominate the high elevation range with the highest salinities. Most other species would be concentrated at mid tidal elevations. Although their elevational distributions overlap, they would exist in a mid tidal mosaic "patchwork" fashion, since the clonal nature of these species means that patches will be monospecific, rather than mixed-species patches.

Restoration and Management Implications

From the survey and literature research it is clear that salt marsh zonation is the result of both competitive displacement and interspecific variation in physiological tolerance. To improve species success in restoration and to emulate the above mentioned pattern observed in natural marshes, initial plantings of each species should be located within the respective optimal range of maximum recorded biomass, provided by the template for revegetation (Fig. 3.13).

Most species will not tolerate salinities above 30 ppt, therefore it is important that regular tidal flushing and freshwater inflows are maintained to prevent hypersalinization.

Salinity and soil texture are interrelated. Silt and clay materials tend to reduce drainage rates and retain more salt than sand (Mitsch and Gooselink, 1993). As the Linwood soils (Chapter 2) are predominantly silt/clay, high salinities may be an on-going problem that needs management. Despite this, the high clay content is desirable in a created wetland likely to be exposed to heavy metal input. Heavy metals are closely adsorbed onto clay particles and immobilised under the near-neutral pH conditions of wetland soils. This is important as although the salt marsh plants tolerated the range of heavy metal levels recorded in "natural marshes", levels recorded in Linwood paddock soils (Chapter 2), are higher than such plants would normally be expected to accommodate. Furthermore, lateral leaching into the Avon-Heathcote Estuary or downward leaching into the groundwater table, will not take place when such soils are reflooded because of the clay subsoil and thus heavy metal levels can be expected to remain high in the long term.

It is also important to maintain a near-neutral soil pH: (i) because an increase in acidity can be fatal for many species; and (ii) an increase in acidity can mobilise heavy metals. Reducing conditions can result from ponding or poor management. Conditions may become so extreme as to kill all or many of the plants, thus forming "bad spots", as described in salt marshes by Chapman (1974).

If restoration is successful, species zonation as determined by the revegetation template should be obvious, and the vegetation cover of each species where present should be approaching 100 %.

4. THE USE OF MESOCOSMS TO DETERMINE METHODS FOR INCREASED RESTORATION SUCCESS.

4.1 Introduction

As the need for wetland restoration and creation becomes more urgent, the need for more effective methods of controlling environmental conditions so that restored systems are successful in supporting the desired biota also increases (Callaway et al., 1997).

Mesocosms (small-scale experimental field systems) are useful for testing restoration techniques and predicting restoration outcomes, before an actual restoration project takes place (Callaway et al., 1997). They can test for a range of factors that may contribute to more successful restoration including planting techniques, substrate conditions, and hydrology. In this experiment mesocosms are used to test the likely success (plant survivorship and growth) of several restoration factors concerning revegetation at the Linwood Paddock site. The treatments tested to determine the survivorship and biomass response are: (i) Linwood restoration site soil compared to natural salt marsh sediment; (ii) salt marsh species; *Juncus maritimus*, *Leptocarpus similis* and *Schoenoplectus pungens*; and (iii) transplant versus nursery stock. Because environmental conditions are controlled in the plots, differences in vegetation between experimental treatments can be readily compared. It was anticipated that any further problems likely to be encountered in the larger scale restoration project would also become apparent.

Restoration experiments in California by Callaway et al. (1997) used tidal mesocosms to determine the effects of different hydrologic treatments on *Salicornia* parameters. However, using tidal mesocosms for restoration purposes is a novel idea in New Zealand. This experiment involves modifying site characteristics and plant source to determine transplant success under specific conditions. Previous studies by Partridge and Wilson (1988a) have also used field transplants. However, these were reciprocal transplants aimed at determining species' tolerance to environmental factors (salinity in particular) and only involved experimental manipulation of species' elevation.

With the costs and uncertainties involved in restoration projects, it is desirable to have a small-scale "trial run" before actual restoration implementation. Mesocosms can also be useful as an on-going research component of larger restoration projects (Adey, 1987). Because of the ability to control some environmental parameters (e.g. substrate type, and plant source) while maintaining seminatural conditions, mesocosms offer great potential for the future evaluation of experimental restoration techniques (Callaway et al, 1997).

4.2 Methods

Study Site

Experimental mesocosms were created within an existing area of salt marsh at Settler's Reserve, Ferrymead (Fig. 4.1). The salt marsh at this site is of very low diversity and consists almost entirely of sea rush *Juncus maritimus*, with only a few upper marsh species (mostly *Sarcocornia quinqueflora*) at higher elevations. The upper marsh extent is limited by light industrial development immediately beyond the marsh. The close proximity of this site to the Linwood Paddocks is advantageous in that it allows for a more accurate application of experimental results to the restoration site. Furthermore, the sparseness of vegetation at this site allows for experimental transplanting without disturbing existing plants.

Mesocosm Description and Treatments

The experiment was a blocked split-plot design, involving three manipulated factors:

1) Substrate (levels: treatment (Linwood paddock soil), control (Heathcote mud).
2) Species (levels: *J. maritimus*, *L. similis*, *S. pungens*).
3) Source (levels: transplant (from Avon marshes), nursery (C.C.C. Linwood nursery)).

Substrate and species treatments were factorially applied at the plot level and the source factor was applied at the subplot level.

A blocked split-plot design was chosen because the three factors were best applied at different scales and then blocked for replication to allow for spatial heterogeneity. This allows for marsh variation especially along elevation, salinity and pH gradients. The experimental configuration was a grid consisting of 36 plots spanning in one dimension, the gradient from low to high tide (Fig. 4.1). All plots were 0.5 x 0.5 m in area and 0.15 m in depth. Plots were marked with unobtrusive wire pegs that minimised tidal erosion at their base. At the beginning of March 1998, all plots were excavated. One half of the plots were filled with topsoil transferred from the Linwood paddocks (with pasture species removed), the other half (control plots) were refilled with Heathcote marsh mud that had been homogenised using a hand trowel. All plots were compacted by foot to obtain a surface elevation consistent with the surrounding marsh to prevent any ponding or increased drainage. Compaction also reduced the loss of sediment in the tidal wash. The mud excavated to create the Linwood soil plots was taken off site, to another part of the salt marsh.

One week after the soil transfer, all plants were transplanted into the plots. Each plot was planted with four individual plants of the same species, two of which were transplanted from the Avon River estuary marshes adjacent to Kibblewhite Street, and two obtained from the City Council's Linwood nursery. To reduce genetic variation among individual plants, each transplant within a plot was obtained by separating the same "parent" plant, and all nursery sourced plants were grown from seed collected from Avon River marshes. Once planted, all plants were trimmed to an equal height of 20 cm. A small sample of untransplanted plants at the Avon River marsh were also trimmed to this height to test for any effects of trimming. A total of 144 "plants" were used to set up the experiment.

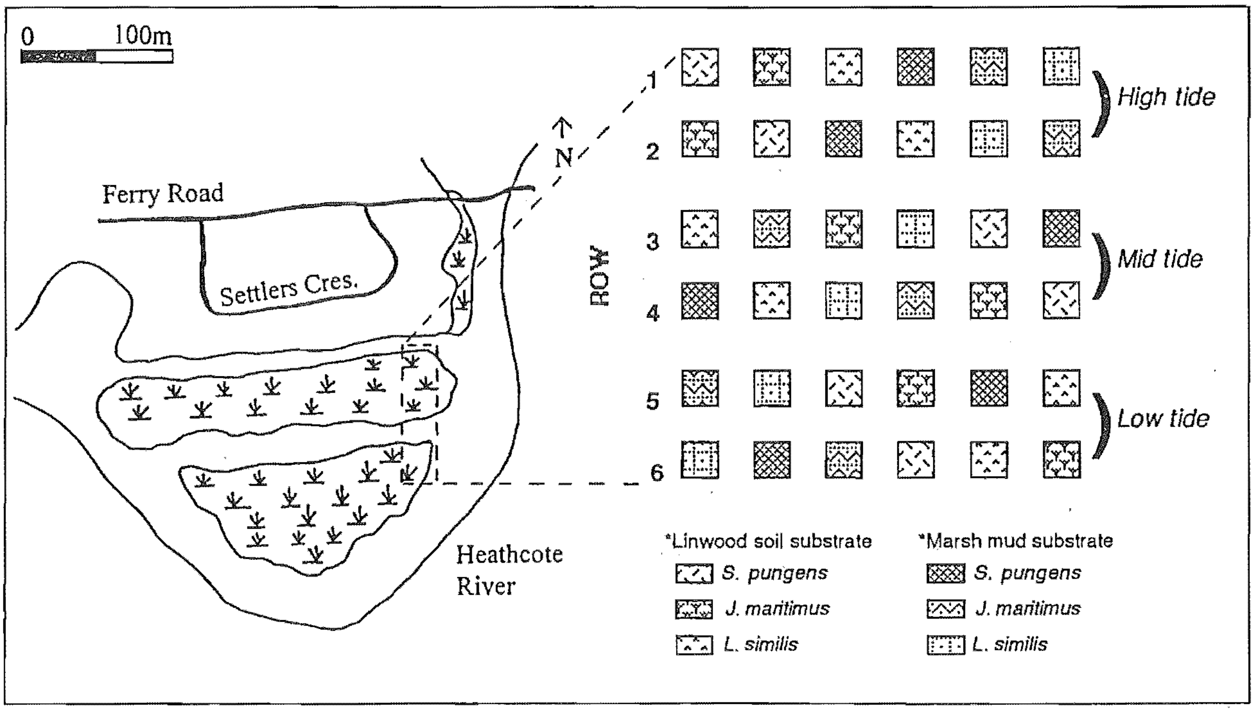


Figure 4.1. Experimental location (left) and layout (right) of tidal mesocosm plots at Settler's Reserve, Ferrymead.

Environmental Conditions

At the time of planting and subsequently at the end of each month until December 1998, soil pH and salinity (ppt) were measured. For this, samples were obtained from the surface (0 - 10 cm) of each individual plot. pH was determined on a 1:1 volume mixture of soil and distilled water using a Metrohm meter. Salinity was determined on a 1:5 volume mixture of soil and distilled water. Using the method of Rhoades and Miyamoto (1990), samples were stirred for 1 minute every 30 minutes and covered with foil to prevent evaporation. Samples were then stored at 4°C for 24 hours and analysed using a hand-held, temperature-compensated refractometer (see Chapter 3 for details).

The particle size distribution and organic content of the Linwood paddock soil and marsh mud used in the experimental plots was also determined at the time of planting. Organic content was determined from loss on ignition at 400°C for 12 hrs and expressed as a percentage of the dry weight. Sediment grain size was determined by dry sieving samples using 0.5, 0.125, and 0.063 mm sieves. Due to the high organic content, samples had to be treated with 4 % HCl and then oven dried before analysis.

Monthly rainfall (mm) was obtained from the Christchurch meteorological service, to correlate with temporal trends in survivorship.

Every two weeks from March to December 1988, plots were checked and cleared of any debris and tidal wrack.

Plant Condition and Growth

Just prior to any spring growth (end of August, 1998) the maximum height of all transplanted individuals was measured. This was repeated again at the end of December, 1998. Although destructive sampling (clipping and weighing) probably provides the most accurate measure of growth (Broome, 1990), in the interests of any long-term studies it was considered too destructive. Therefore, indirect estimates of standing biomass were made from predictive equations, derived by linear regression relating shoot length of plants from existing salt marshes, to dry weight biomass. Similar equations have been derived and used by Brownlow (1997) and Hocking (1989), for *Phragmites* in inland Australia. To determine aerial dry weight, plants of all three test species from the Avon and Heathcote salt marshes were clipped at the sediment surface, dried in a oven at 60°C and weighed to the nearest gram. Linear and log-linear forms were tested for each species and the most appropriate one used for each.

Statistical Analyses

Mortality was tested using a logistic analysis of variance (ANOVA) (although it could not test the split-plot effect properly, except in having the subplot effect and its interactions tested last). An ANOVA revised for the split-plot design, with interaction (species, substrate, source, and their interaction) was used to analyse the plant biomass results from December. Data from living plants only was included in this analysis.

4.3 Results

Environmental Conditions

Despite regular tidal flushing at the beginning of the experiment, footprints associated with the experimental set-up, took until the end of June to disappear and wheelbarrow tyre marks took until the end of September.

Mean plot salinities ranged from 15.77 to 55.78 ppt. Mesocosm soils were flooded twice daily by the tide throughout the year except in September and November, when lack of rainfall lowered river levels preventing tidal innundation. Consequently, tidal waters failed to reach plots for 10 days in September, and 14 days in November. In these instances only row 6 (nearest the active channel) remained moist. In all remaining plots the substrate was dry and cracked, with salt clearly visible on the substrate surface. This resulted in elevated soil salinities for these two months (Fig. 4.2). Smaller fluctuations in rainfall, although not affecting tidal flushing, also negatively influenced soil salinities (Fig. 4.2). There was little difference in soil salinity between both substrate types from March until August (Fig. 4.2). From September to November Linwood soil salinities were on average 10 ppt higher than those of the native marsh mud (Fig. 4.2).

There was an initial pH difference between the two substrate types (Fig. 4.3), but by May there was a general convergence toward pH 7. From June to December levels fluctuated around pH 7, with both the estuarine mud and Linwood paddock soil following the same trend. In November there was a drop in pH, corresponding to lowered rainfall prior to measurement, however levels returned to near neutral in December (Fig. 4.3). All pH levels were within the range from 5.84 to 7.42.

Linwood paddock soil used in the plots was much higher in silt/clay content than the marsh mud, which had similar proportions of silt/clay and fine-medium sand (Table

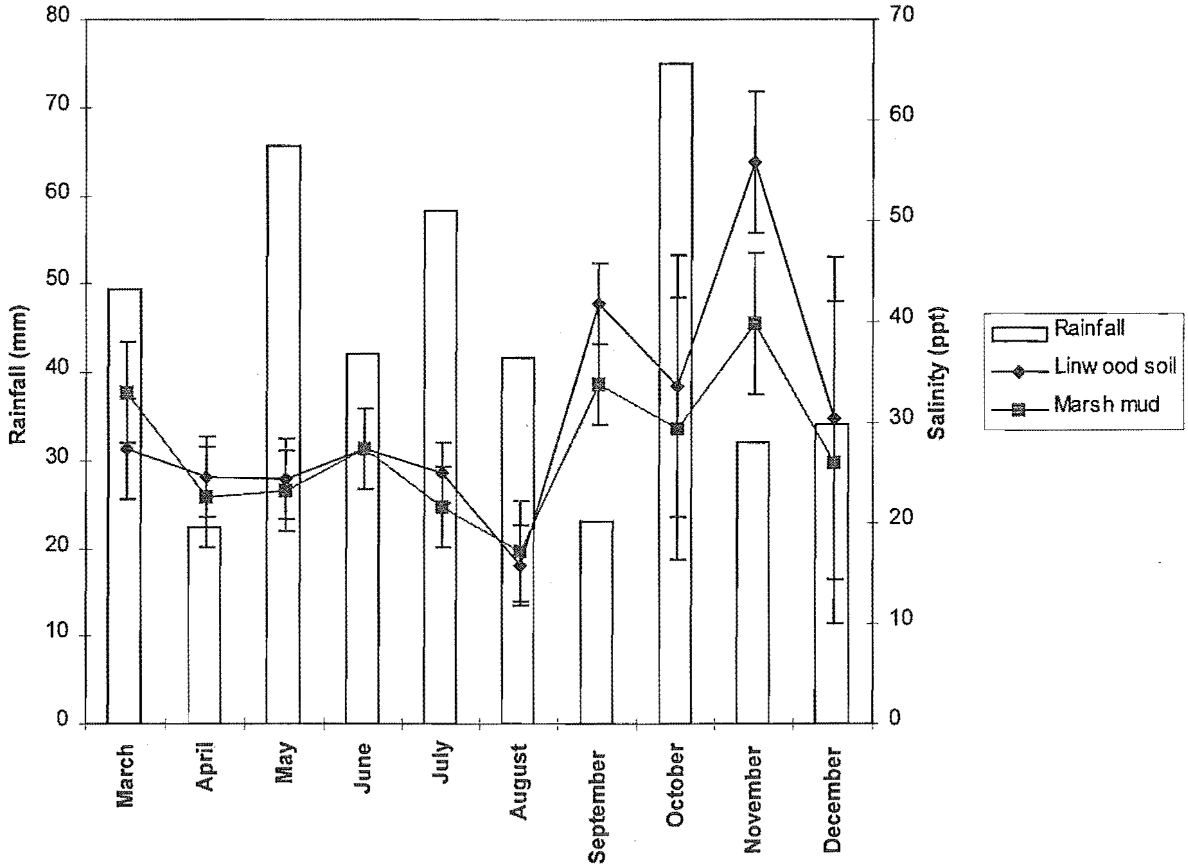


Figure 4.2. Mean salinity (ppt) (± S.D.) of native marsh mud and Linwood paddock soil in experimental plots and monthly rainfall totals (mm).

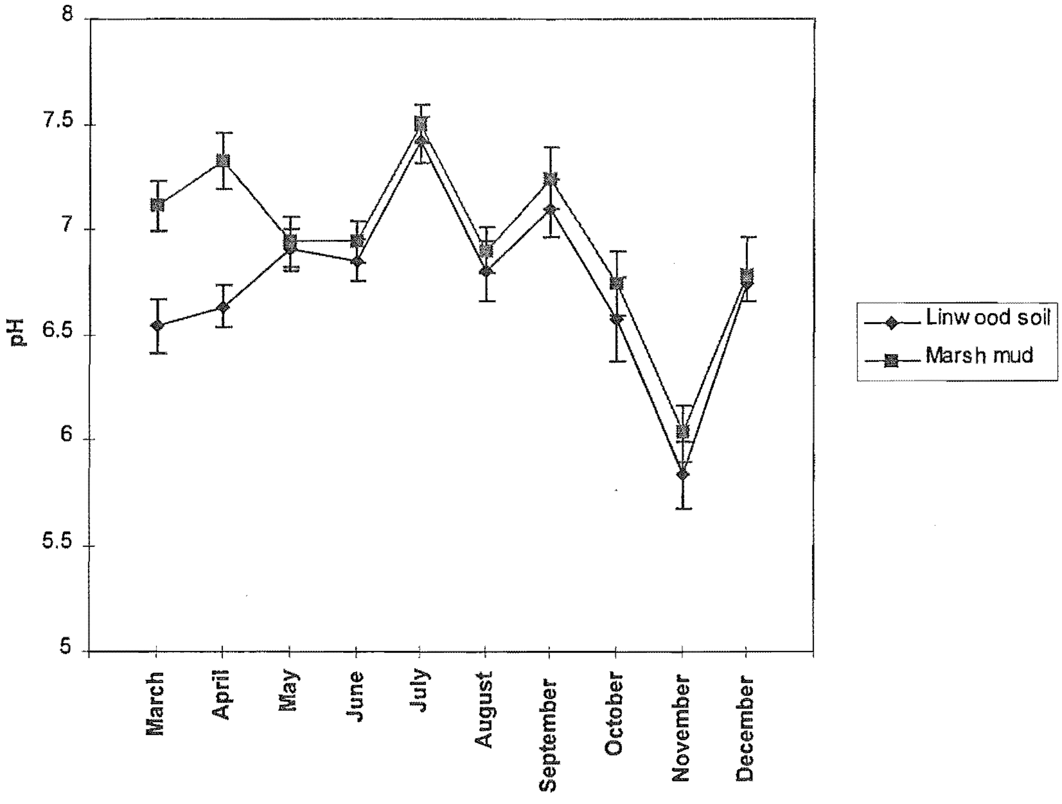


Figure 4.3. Mean pH (± S.D.) of native marsh mud and Linwood paddock soil in experimental plots.

4.1). Hence the lower pH and higher salinity observed in Linwood soils in periods of drought. The organic component of the two was similar.

Table 4.1. Percentage particle size distribution of sediment core samples collected from Linwood paddock soil and Settlers Reserve marsh mud, March 1998.

Substrate	Marsh	Linwood
Particle Size	%	%
> 0.500 mm	0.47	1.48
Coarse sand 0.500 - 0.125 mm	42.68	1.69
Fine – medium sand 0.125 - 0.063 mm	1.85	0.82
Very fine sand < 0.063 mm	46.27	83.74
Silt/clay Organic component	8.88	11.40

Plant Condition and Growth

All plants survived and appeared healthy one month after planting. By the end of May all of the annual *Schoenoplectus pungens* plants appeared to have died back in a normal seasonal manner; however, all failed to regenerate in spring, thus they had actually died immediately prior to or during dormancy. All of the perennial *Juncus maritimus* and *Leptocarpus similis* plants were still alive at the end of May. In August there was still 100 % survival for the *Leptocarpus similis* plants but the survival rate for *Juncus maritimus* had decreased to 87.5 % (Table 4.2). The increased salinities in September resulted in most *Juncus maritimus* and *Leptocarpus similis* plants appearing stressed (chlorotic shoots, wilting) from this date. In December the percent survival for *Leptocarpus similis* remained at 100 %, but *Juncus maritimus* survival had decreased a further 14.5 % (Table 4.3). The analysis showed that the difference in survival between *Juncus maritimus* and *Leptocarpus similis* was significant ($F = 13.48$, $p < 0.05$). The source of the plants was also significant ($F = 8.97$, $p < 0.05$), with nursery-sourced *Juncus maritimus* plants having a higher mortality rate than those sourced from natural stocks.

For *Juncus maritimus* in the Canterbury region, the relationship between shoot length (x) and dry weight biomass (y) is of the form:

$$\log y = a + bx \tag{4.1}$$

In this study, the biomass of *Juncus maritimus* in experimental plots was indirectly estimated using Equation 4.1, using the derived parameter values $a = -1.40$, $b = 0.02$. The line described by these parameter values is shown in Figure 4.4.

For *Leptocarpus similis* in the Canterbury region, the relationship between shoot length (x) and dry weight biomass (y) is of the form:

$$y = a + bx \tag{4.2}$$

In this study, the biomass of *Leptocarpus similis* in experimental plots was indirectly estimated using Equation 4.2, using the derived parameter values $a = 0.03$, $b = 0.008$. The line described by these parameter values is shown in Figure 4.5.

Height measurements and subsequent plant biomass estimates in August (Table 4.2) and December (Table 4.3) revealed that there was a significant difference in biomass between the two species (*Juncus* and *Leptocarpus*) (Table 4.4). There was also a significant difference in biomass between plants sourced from the nursery and those sourced from natural marsh areas (Table 4.4). *Leptocarpus similis* has significantly greater biomass and survival rate than *Juncus maritimus*, and plants sourced from natural marsh areas have significantly greater biomass than those propagated in a nursery. The substrate had no significant effect on growth for either species or source of species (Table 4.4). In addition, the mean height of trimmed plants from Avon marshes in December, was 23.5 and 24.77 cm for *Leptocarpus similis* and *Juncus maritimus* respectively. As these heights are greater than the average measured in the experiment, trimming the experimental plants does not appear to be a cause for their retarded growth.

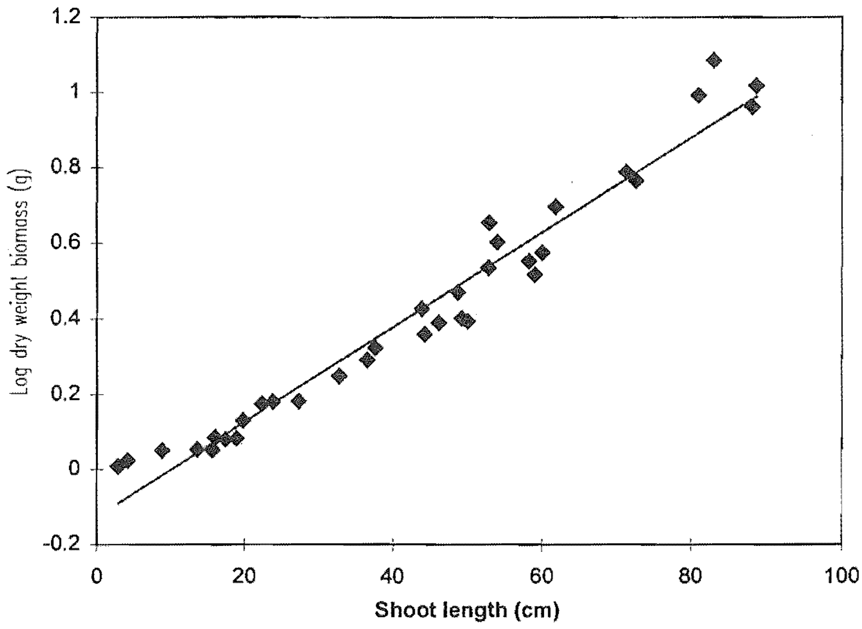


Figure 4.4. Relationship of *Juncus maritimus* shoot length to dry weight biomass.

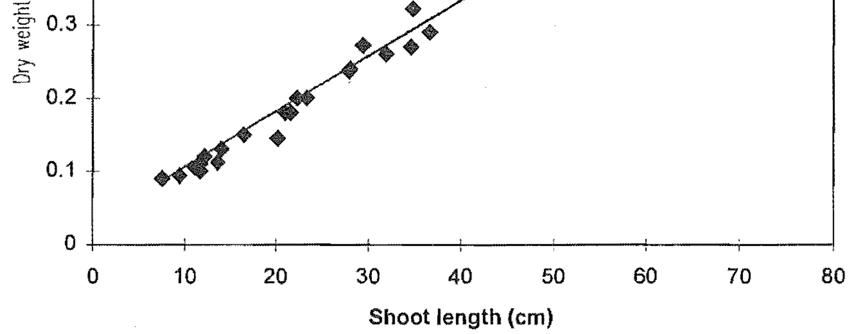


Figure 4.5. Relationship of *Leptocarpus similis* shoot length to dry weight biomass.

Table 4.2. Percentage survival and biomass (g), of plants in experimental tidal plots, August 1998*

	Juncus	*Leptocarpus*	*Schoenoplectus*
% survival	87.5	100	0
Overall biomass	0.09 (0.04)	0.19 (0.04)	-
Nursery source	0.07 (0.06)	0.18 (0.07)	-
Natural source	0.12 (0.05)	0.20 (0.05)	-
Paddock soil	0.10 (0.06)	0.19 (0.06)	-
Marsh mud	0.09 (0.06)	0.18 (0.06)	-

*Values are means for each treatment (±SD).

Table 4.3. Percentage survival and biomass (g), of plants in experimental tidal plots, December 1998*

	Juncus	*Leptocarpus*	*Schoenoplectus*
% survival	73.0	100	0
Overall biomass	0.01 (0.07)	0.19 (0.06)	-
Nursery source	0.05 (0.05)	0.18 (0.07)	-
Natural source	0.13 (0.05)	0.26 (0.06)	-
Paddock soil	0.10 (0.07)	0.19 (0.06)	-
Marsh mud	0.04 (0.15)	0.18 (0.06)	-

*Values are means for each treatment (±SD).

Table 4.4. Analysis of variance (ANOVA) for *Juncus maritimus* and *Leptocarpus similis* biomass, December 1998.

	Df	Sum of Sq	Mean Sq	F Value	Pr(F)
Row	5	0.0235	0.0047	3.0529	0.0426
Species	1	0.1243	0.1243	80.8469	0.0000
Substrate	1	0.0045	0.0045	2.9084	0.1087
Species:Substrate	1	0.0002	0.0002	0.1328	0.7206
Plot Error	15	0.0231	0.0015		
Source	1	0.0204	0.0204	26.3833	0.0000
Source:Species	1	0.0147	0.0147	18.9839	0.0001
Source:Substrate	1	0.0000	0.0000	0.0594	0.8084
Source:Species:Substrate	1	0.0001	0.0001	0.1547	0.6956
Subplot Error	54	0.0418	0.0008		

4.4 Discussion

Experimental mesocosms experienced extremes of dryness and salinity identical to those in surrounding natural salt marsh areas. They were also exposed to natural levels of herbivory and tidal stress. The spatial and temporal scale of this experiment appeared sufficient to reflect 'restored' natural conditions whilst still maintaining a degree of environmental control. Larger mesocosms may predict restoration outcomes more accurately, but the results will be more variable.

Environmental Conditions

The trend towards neutrality of the slightly acidic Linwood paddock soil is consistent with what is generally expected when permanently drained land becomes flooded, as occurs in the construction of wetlands. When previously drained soils are flooded (as would be the case if the Linwood Paddocks were reflooded), alkaline soils generally decrease in pH and acid soils increase in pH (Mitsch and Gooselink, 1993) - the former resulting from the build up of CO_2 and then carbonic acid, the latter stemming from the reduction of ferric iron hydroxides (Mitsch and Gooselink, 1993). The subsequent "neutral" pH is generally in the range from 6.7 - 7.2 (Mitsch and

Gooselink, 1993), which is slightly narrower than the range measured for both marsh mud and Linwood soil in this experiment.

Soil water salinity is a dominant factor in the productivity and species selection of the salt marsh (Ranwell, 1972; Chapman, 1974). Frequent rainfall tends to decrease salinities, whilst frequent periods of drought lead to higher salt concentrations. A severe lack of rainfall will reduce tidal flushing and further increase salinities to a point where some plants may die, due to associated physiological stresses. Certainly, this appeared to be the case in September and November when lack of sufficient rainfall reduced tidal flooding and lead to extreme soil salinities, leaving most plants in a generally stressed condition. Soil texture also influences salt concentrations. Silt and clay materials tend to reduce drainage rates and retain more salt than sand (Mitsch and Gooselink, 1993). Linwood soil has a higher silt and clay content than the marsh mud used in the experimental plots. Consequently, although the trend in salt concentrations was identical once flooded, those plots that contained Linwood soil maintained a higher salt concentration than those with marsh mud. All mean soil salinities were within the range of those found for fully tidal experimental mesocosms by (Callaway et al., 1997). However, some of the more extreme salinities experienced in November, e.g. 76, 84 & 102 ppt, were more similar to experimental mesocosms studied by Callaway et al. (1997), where the tidal influence had been impounded or excluded.

Both the marsh mud and Linwood substrate are classed as organic soils (>5% organic matter). This means they have a higher cation exchange capacity than mineral soils, due to their domination by the hydrogen ion (Mitsch and Gooselink, 1993). Organic soils can therefore remove some contaminants through ion exchange and can enhance nitrogen removal by providing an energy source and anaerobic conditions appropriate for denitrification (Mitsch and Gooselink, 1993). Organic matter in salt marsh soils generally varies between 8 and 25 percent (Long and Mason, 1983); the level determined in both experimental substrates is at the bottom of this range, but the observed lower level is not unexpected in such a "new" marsh. Once the vegetation establishes the substrate organic content would be expected to increase accordingly.

The considerable length of time that it took for any footprints or tyre tracks to disappear in the marsh, is an important point to note in when working in such areas. Not only are such tracks unsightly, but they increase ponding and inhibit plant establishment in some areas. Tracks can be minimised by working at low tide and by using the same track each time.

Plant Growth and Condition

Plants showed similar survival in natural sediment and potentially contaminated Linwood Paddock soil. Therefore, the substrate appears to have no significant effect on plant growth in the experiment. Indeed, salt marsh sediment composition does not seem to be a critical factor in colonisation by plants unless the substrate is almost pure sand that is subject to rapid desiccation at the upper elevations (Mitsch and Gooselink, 1993).

However, there were significant differences in survival and biomass between the two plant sources. The death of nursery sourced plants was most likely caused by a weakening of the plant structure through herbivory or tidal stress, followed by subsequent removal with the tidal wash. Indeed, the majority of dead or lost plants exhibited signs of rabbit grazing prior to their demise. A rabbit problem has been noted in the Ferrymead area, and other City Council plantings in the vicinity have had rabbit control (Dennis Preston, C.C.C., pers. comm.). The significant difference between the two plant sources appears to be caused by the nursery-sourced plants being softer in stem than the hardier natural stocks. Consequently they are more susceptible to herbivory and tidal stress, hence their lower survival rate for *Juncus maritimus*, and biomass for both *Juncus maritimus* and *Leptocarpus similis*. The difference in survival between *Juncus maritimus* and *Leptocarpus similis* was evident from August (Table 4.2), and even more obvious in December (Table 4.3). This is consistent with the pattern observed in natural marshes (Chapter 3), where *Leptocarpus similis* is the competitive dominant where the two species coexist. However, it was not entirely expected in this marsh where *Juncus maritimus* grows in extensive monocultural stands and *Leptocarpus similis* does not naturally exist at all.

The complete failure of *Schoenoplectus pungens* may be due to the time of planting. Perhaps spring planting would have facilitated better root establishment. However, Partridge and Wilson (1988a) also found that *Schoenoplectus pungens* and *Juncus maritimus* did not transplant well, even in reciprocal transplants were only elevation was manipulated manually. They concluded that this was probably as a result of damage to the plants during transplanting. Furthermore, Callaway et al. (1997) state that although it may be possible to establish plants from seedlings or cuttings, it is questionable whether or not these individuals will be successful in regenerating. If future generations cannot establish easily, the population will not be self-sustaining (Callaway et al., 1997). This issue may not be a concern for common, clonal species such as *Juncus maritimus* and *Leptocarpus similis*, but obviously it must be addressed for annuals and short-lived perennials such *Schoenoplectus pungens*.

Restoration Implications

Experimental manipulation and mesocosms allow for the replication of restoration techniques, the combination of multiple factors, and the statistical analysis of cause and effect relationships, something that has been missing from past restoration activities (Kondolf, 1995).

This study highlighted the difference between plant tolerance to conditions for persistence, and conditions for establishment. Initially, annual *Schoenoplectus pungens* plants survived in trial plots, but they failed to regenerate in spring following seasonal die-back. Salinity levels were elevated in September when regeneration was expected, but they were still well within those experienced by established *Schoenplectus pungens* in natural marsh areas (Chapter 3). It is probable that *Schoenplectus pungens* tolerance of environmental conditions is much narrower for establishment than for persistence. This has been shown for several other species e.g. *Salicornia virginica* (Beare and Zedler, 1987; Callaway et al., 1997) and *Salicornia subterminalis* (Kuhn and Zedler, 1997). The implication of this is that restoration designs should be focussed on providing the conditions necessary for plant establishment rather than only persistence.

Another major factor in restoration success is the source of plants. Currently all City Council revegetation projects utilise plants sourced from nursery stock, because transplanting from natural marsh areas is considered too destructive at the donor site. Although nursery stock is grown from seed collected from natural marshes, the artificial nursery environment means that nursery stocks are not as hardy as natural stocks. Perhaps once germinated, the nursery stocks could be transplanted to a protected marsh area, so that the majority of plant development takes place under conditions analogous to those in the proposed restoration site. In addition, it is surprising how many individual plants can be separated from one parent plant. Therefore, transplanting from natural marsh areas does not require the removal of large amounts of vegetation and may not be as destructive as first thought.

Herbivory, especially by rabbits at low tide, is another problem to be controlled for. Rabbit-proof fencing and repellent have been used successfully in other revegetation attempts by the City Council and should continue to be used in future restoration projects.

The experimental substrates appeared to have no effect on plant growth or biomass in this study. However, the higher salt concentrations recorded in the Linwood soil due to increased salt retention in the predominantly silt/clay substrate, may have negative effects on plant growth in the longer-term, or in more extreme seasons with longer periods of drought. During salt marsh restoration, excessive salinities may have to be controlled for in both design (involving application of topsoil with a lower clay content) and management (including increased watering in times of drought). Otherwise, the use of Linwood soil as a salt marsh substrate does not appear to affect marsh soil chemistry or present an impediment to restoration plant growth.

5. ASSESSMENT OF PREVIOUS WETLAND RESTORATION AND CREATION PROJECTS

5.1 Introduction

Restoration success depends on the long-term ability to manage and protect wetlands and adjacent buffer areas. As wetland losses continue and restoration and creation efforts increase, it is apparent that long-term success can be quite different from short-term success. Revegetation of a restored or created wetland over a relatively short period of time (e.g. one year) is no guarantee that the area will continue to function over longer periods of time (e.g. decades) (Kusler and Kentula, 1990). Long-term research data from Canterbury can be used in determining strategies for restored salt marsh implementation at the Linwood Paddocks and elsewhere in New Zealand and the temperate world. It is essential to understanding the impacts of wetland management decisions and redirecting them where appropriate.

This chapter treats existing Christchurch City Council revegetation projects as experiments in progress, in an attempt to learn from what has already been done and to use that information to improve future restoration design, implementation and management.

The assessment includes the restored Charlesworth Street Reserve, the bankworks on the margins of Oxidation Ponds 5 and 6 of the Bromley Sewage Treatment Works, and the bankworks at Devil's Elbow on the lower Heathcote River margins. All projects are in close proximity to the proposed Linwood Paddock restoration site. Therefore, results should be relevant and transferable to the restoration plan. A range of designs and implementation strategies have been used, thus allowing an evaluation of techniques. Although all projects were not designed primarily to facilitate salt marsh development, salt marsh species have been used in revegetating these tidal environments. Regardless of whether the project aims were ecosystem rehabilitation, bank stabilisation or aesthetic appeal, assessments of plant growth and condition as a result of regulatory decisions in this habitat are still relevant. Indeed, the information from these projects is the only long-term data obtainable in the Avon-Heathcote area.

5.2 Methods

Study Sites

Plant condition assessment was carried out at four revegetated sites surrounding the Avon-Heathcote Estuary. The projects assessed were; the Avon-Heathcote Estuary frontage stabilisation trials along the margins of Oxidation Pond Nos. 5 and 6 of the Bromley Sewage Treatment Works, the Charlesworth Street Reserve adjacent to Humphreys Drive, and the Lower Heathcote bankworks, Devil's Elbow (Fig. 5.1). The planting regime and implementation strategies for each project are detailed in Figures 5.2 - 5.9.

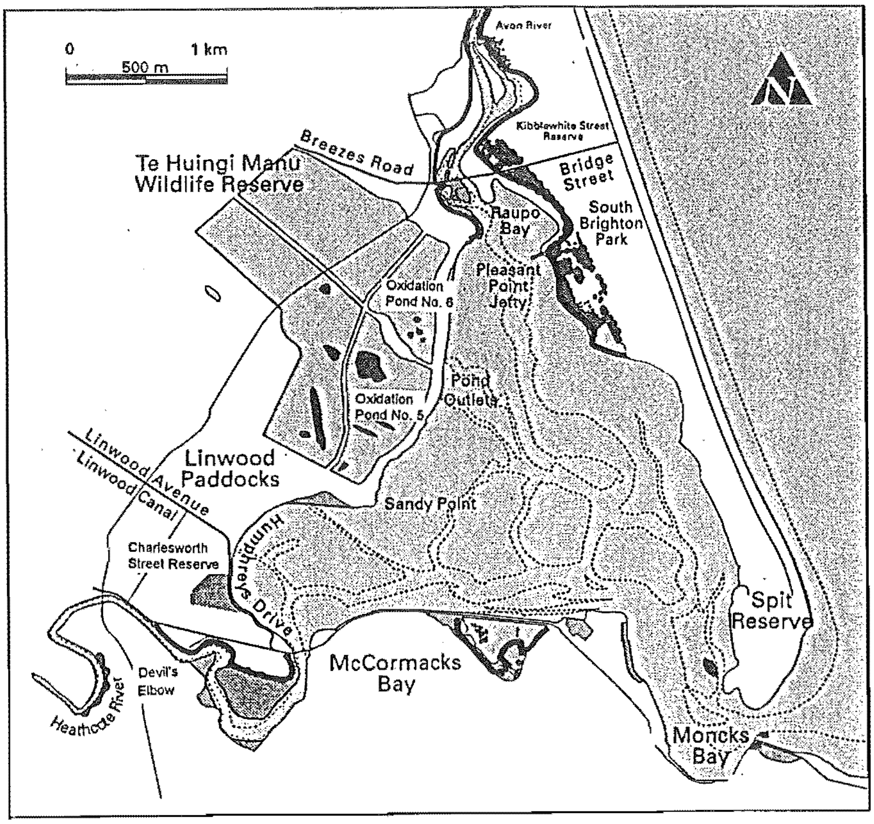


Figure 5.1. Location map of all revegetation projects assessed in proximity to the Linwood Paddocks (adapted from Owen, 1992).

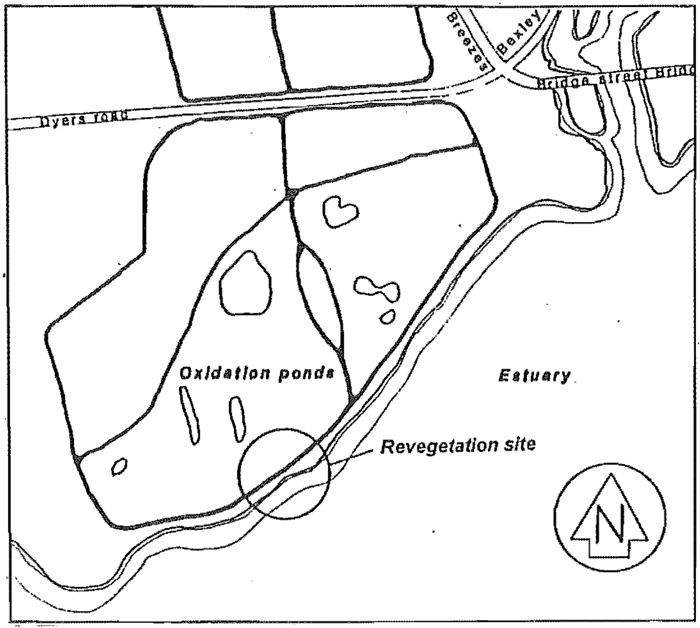
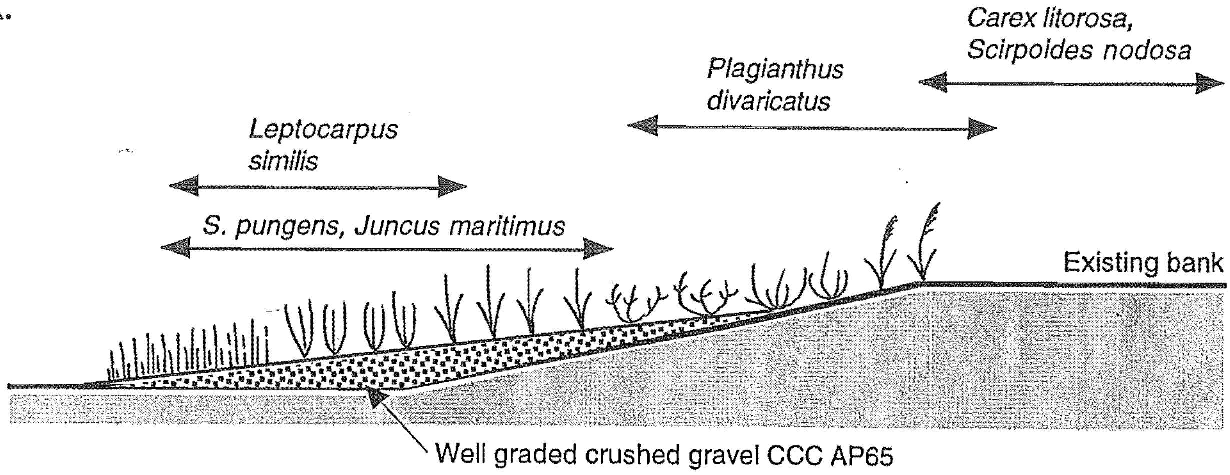


Figure 5.2. Revegetation site along the margin of Oxidation Pond No. 5 and the Avon-Heathcote Estuary.

Design and implementation:

- The purpose of revegetation adjacent to Oxidation Pond No. 5 was to provide 80 m of bank protection.
- The existing concrete rubble was replaced with 2 grades of gravel trialed to assess their effectiveness against erosion generated by easterly swells.
- The gradient of all works is less than 1:6 and provides a gradual transition from the estuarine mud flats.
- All plant material was nursery sourced, germinated from local stocks.
- Plants were planted in large monospecific patches according to the zones depicted below. Each plant was set 50 mm into the gravel substrate.
- After planting, all plants were treated with Mesurol bird repellent.

A.



B.

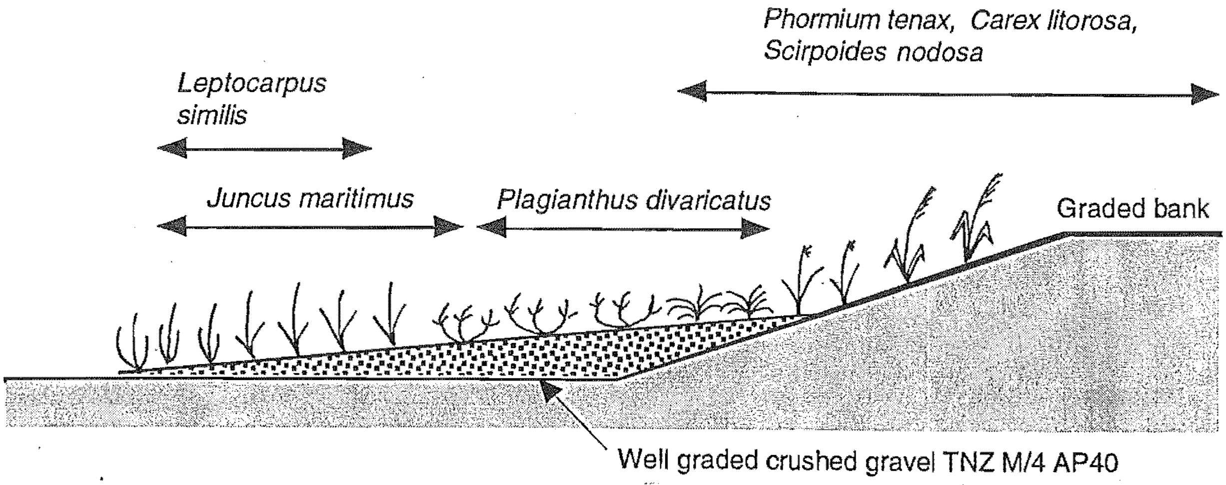


Figure 5.3. Zonation of species used in revegetation at Oxidation Pond No. 5. (A) depicts the zonation when the gradient was 1:6, (B) depicts the zonation when the gradient was 1:8. Diagrams have been constructed using information provided by Dennis Preston, City Design, Christchurch City Council. This design was implemented in 1995.

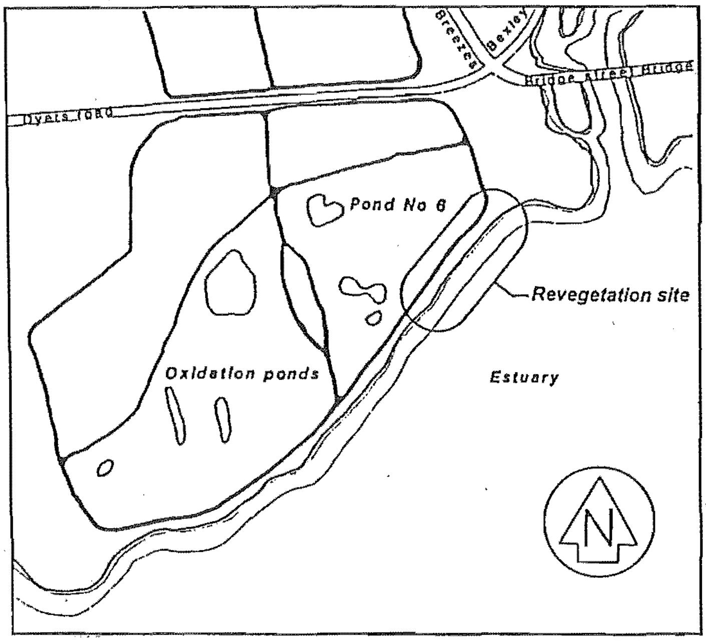


Figure 5.4. Revegetation site along the margin of Oxidation Pond No. 6 and the Avon-Heathcote Estuary.

Design and implementation:

- The purpose of revegetation adjacent to Oxidation Pond No. 6 was to provide 380m of bank protection.
- The existing concrete rubble was replaced with a well graded crushes gravel.
- The gradient of all works is less than 1:6 and provides a gradual transition from the estuarine mud flats.
- All plant material was nursery sourced, germinated from local stocks.
- Plants were planted in large monospecific patches according to the zones depicted below. Each plant was set 50 mm into the gravel substrate.
- After planting, all plants were treated with Mesurol bird repellent.

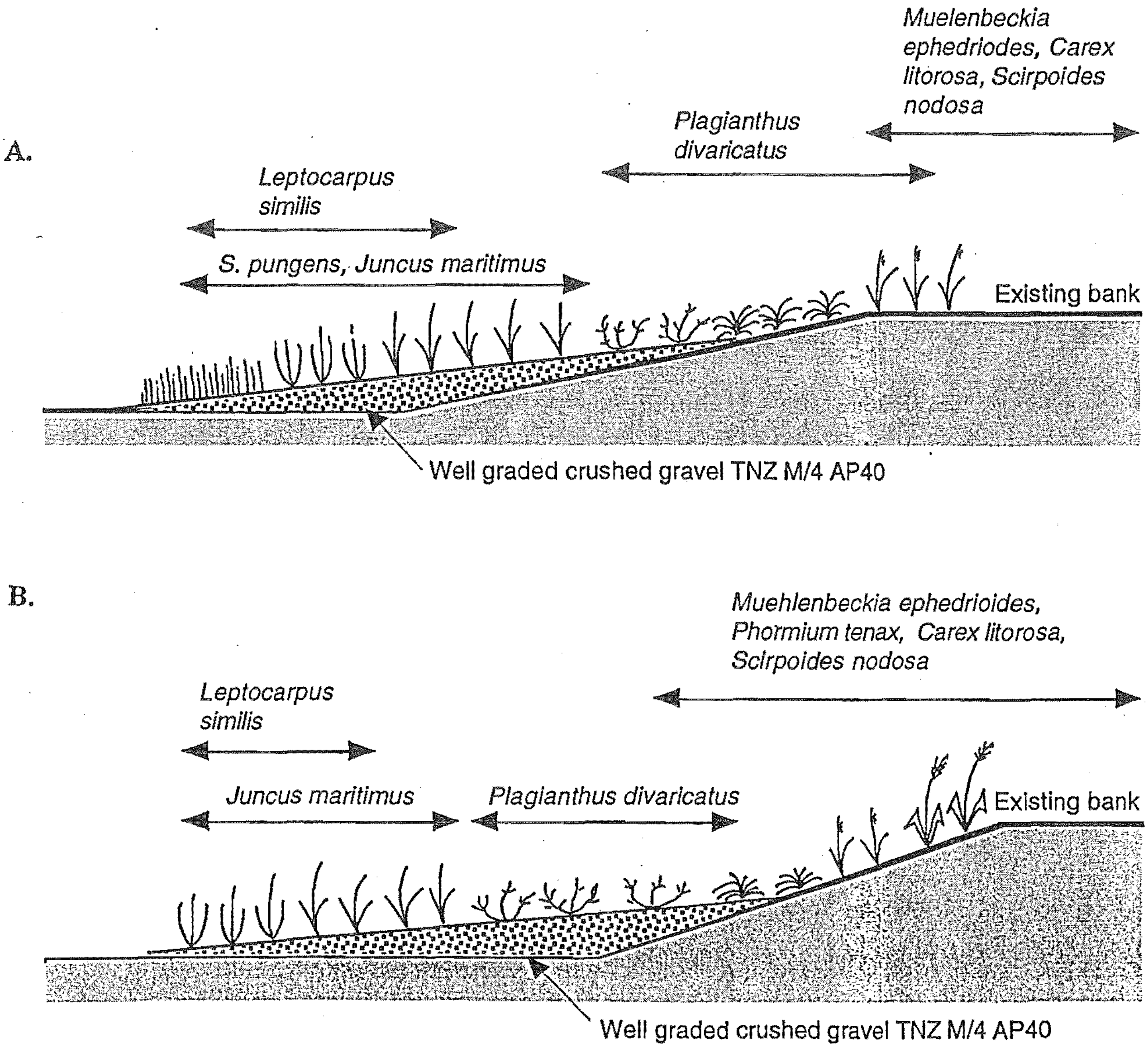


Figure 5.5. Zonation of species used in revegetation at Oxidation Pond No. 6. (A) depicts the zonation when the gradient was 1:6, (B) depicts the zonation when the gradient was 1:8. Diagrams have been constructed using information provided by Dennis Preston, City Design, Christchurch City Council. This design was implemented in 1997.

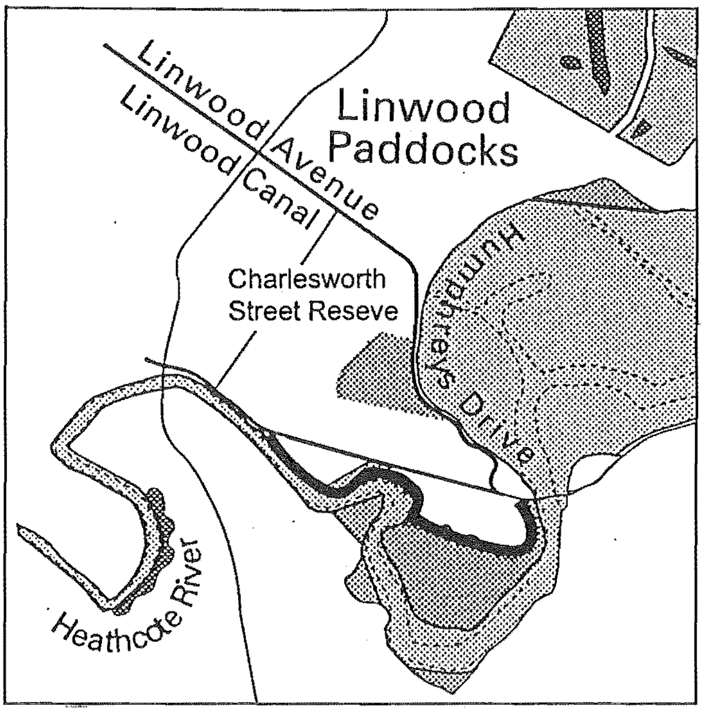


Figure 5.6. Revegetation site at the Charlesworth Street Reserve adjacent to the Avon-Heathcote Estuary (adapted from Owen, 1992).

Design and implementation:

- The purpose of revegetation was facilitation of a natural salt marsh.
- The existing topsoil was scraped and despite extensive design plans, only a few *Juncus maritimus, Leptocarpus similis* and *Mimulus repens* were planted initially. Other planting followed according to the zones depicted below.
- All plant material was nursery sourced, germinated from local stocks.
- Strong grass growth was predicted and blanket spraying was implemented before planting.
- Unlike other sites, this reserve relies on a single culvert to provide adequate tidal influence and flushing.

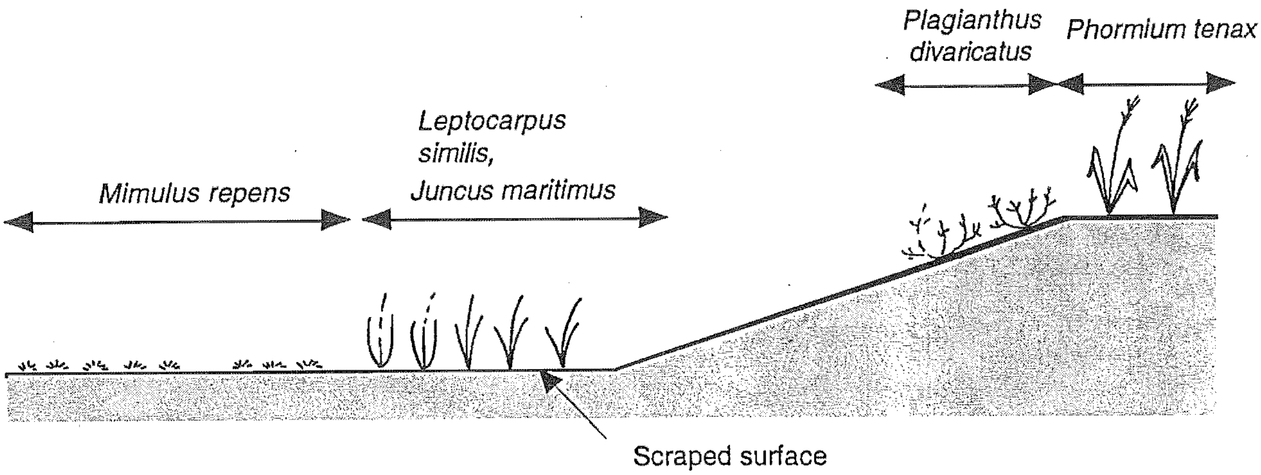


Figure 5.7. Zonation of species used in revegetation at the Charlesworth Street Reserve. Although other species have colonised this site, only those planted have been included. Diagrams have been constructed using information provided by Dennis Preston, City Design, Christchurch City Council. This design was implemented in 1994.

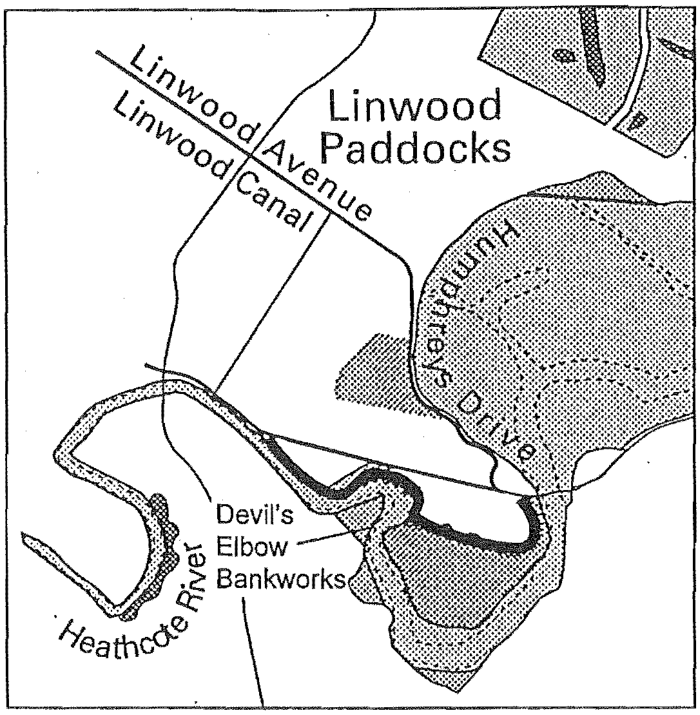


Figure 5.8. Revegetation site bordering Devil's Elbow (lower Heathcote River) (adapted from Owen, 1992).

Design and implementation:

- The purpose of revegetation surrounding the Devils Elbow section of the lower Heathcote Loop was bank stabilisation and creation of an aesthetic area for workers in the adjacent light industrial area.
- The existing bank was replaced with fine crushed gravel and covered with 'Enkamat' through which the plants were planted.
- All plant material was nursery sourced, germinated from local stocks.
- Plants were planted in informal bands according to the zones depicted below. Each plant was set 50 mm into the gravel substrate.
- Plants were coated with an egg and resin mix to deter rabbits.

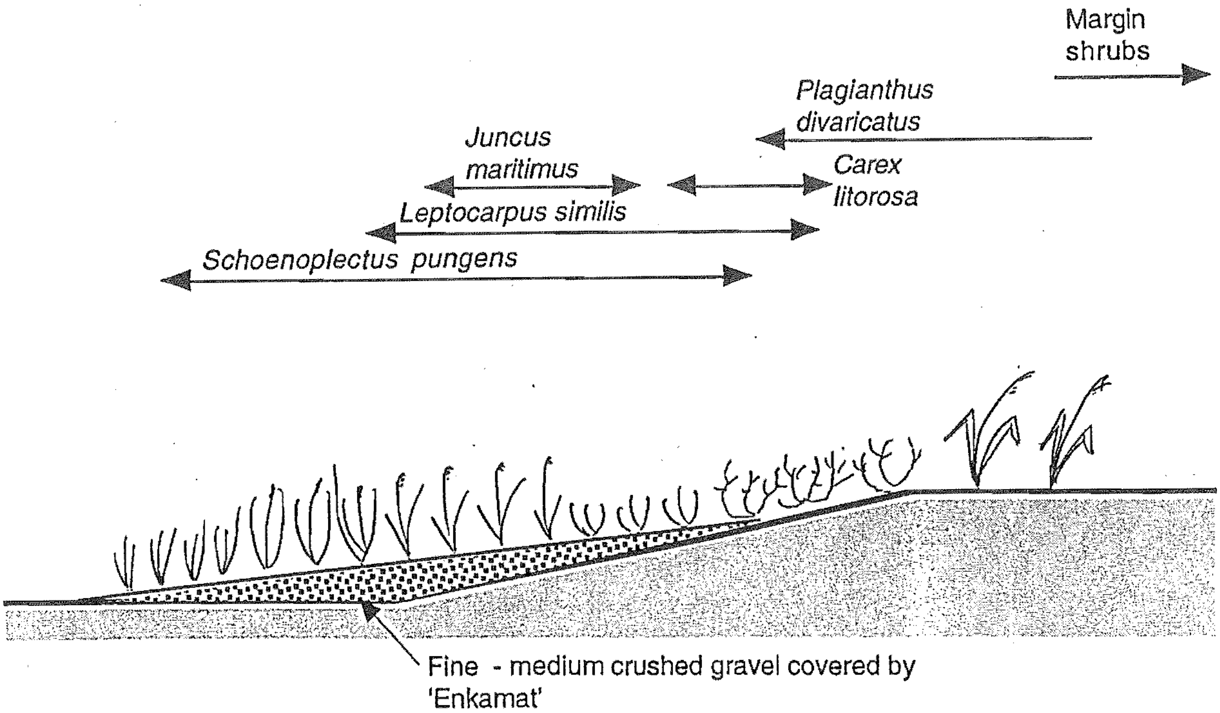


Figure 5.9. Zonation of species used in revegetation at the Devil's Elbow bankworks. Squares were cut in the 'Enkamat' for planting purposes. Diagrams have been constructed using information provided by Dennis Preston, City Design, Christchurch City Council. This design was implemented in 1995.

Plant Growth and Condition Assessment

Stratified sampling was used to determine the relative success of species used in revegetation attempts in August and December 1998. Using the methodology of Erwin (1990), project sites were separated into three tidal zones (low, mid and high). Seven $1m^2$ quadrats were then randomly allocated and assessed within each zone. The occurrence, number, height and condition of each species were recorded in each quadrat. Due to the variable nature of project design and implementation, only qualitative measurement parameters and criteria for plant condition assessment adapted from Erwin (1990) were used to compare and evaluate data from different projects. The following criteria were ranked to enable site and species scoring. A higher number indicates a better-conditioned species or site.

Criteria:	*Score:*
Live	5
Tip die back	4
Basal sprouts only	3
Stressed	2
Apparently dead	1
Not found	0
Dead	0

Live

Plant appears in apparently good condition, leaves green, no symptoms of wilting, die-back, or chlorotic appearance of leaves.

Tip die back

The main stem is in good condition, but the most apical portions are in very poor condition exhibiting wilting and die-back symptoms.

Basal sprouts only

The main stem is dead but new growth is initiated from the stem base or the root stock.

Stressed

Plant appears to be in a generally poor condition - chlorotic leaves, wilting and leaf drop.

Apparently dead

The general plant appearance is dry and brittle with no live tissue observed and there is no observable green foliage growth.

Not found

In some cases plants are not found during a particular sampling period. If a plant is not found on two successive sampling periods, it was counted as dead.

Dead

A decision as to whether a plant was dead was generally made only following a sampling period in which the plant was classified as "apparently dead". Only if initial observations indicated that the stem was in such poor condition that survival was unlikely should a plant be listed as dead.

5.3 Results

Plant Occurrence

Scirpoides nodosa is the only species to survive at all sites where it was planted (Tables 5.1 and 5.2). *Juncus maritimus* and *Phormium tenax* were planted at all sites and both species only failed to survive at Oxidation Pond No. 5 (Table 5.1). *Leptocarpus similis* was also planted at all sites and failed to survive only at the Charlesworth Street Reserve (Table 5.3). *Schoenoplectus pungens* failed to survive at all sites where it was planted (Tables 5.1, 5.2 and 5.4). This is the only species to go extinct following planting at the Devil's Elbow bankworks (Table 5.4).

Species not originally planted have colonised successfully without intervention only at the Devil's Elbow bankworks and the Charlesworth Street Reserve (Tables 5.3 and 5.4). The main difference between these two sites and the Oxidation Pond sites appears to be the substrate type, which is relatively finer at the Devil's Elbow bankworks and the Charlesworth Street Reserve. Self-colonising species include *Sarcocornia quinqueflora*, *Spergularia media*, *Selliera radicans* and *Suaeda novae-zealandiae*. *S. quinqueflora* is the only species to have successfully self-colonised two sites.

Table 5.1. Plant occurrence and source at Oxidation Pond No. 5.

Species planted	Species planted and survived	Species self-colonised
Juncus maritimus	*Leptocarpus similis*	
Leptocarpus similis	*Scirpoides nodosa*	
Schoenoplectus pungens	*Carex litorosa*	
Plagianthus divaricatus		
Carex litorosa		
Scirpoides nodosa		
Phormium tenax		

Table 5.2. Plant occurrence and source at Oxidation Pond No. 6.

Species planted	Species planted and survived	Species self-colonised
Juncus maritimus	*Juncus maritimus*	
Leptocarpus similis	*Leptocarpus similis*	
Schoenoplectus pungens	*Scirpoides nodosa*	
Plagianthus divaricatus	*Phormium tenax*	
Carex litorosa		
Scirpoides nodosa		
Muelenbeckia ephedriodes		
Phormium tenax		

Table 5.3. Plant occurrence and source at the Charlesworth Street Reserve.

Species planted	Species planted and survived	Species self-colonised
Juncus maritimus	*Juncus maritimus*	*Sarcocornia quinqueflora*
Leptocarpus similis	*Plagianthus divaricatus*	*Spergularia media*
Plagianthus divaricatus	*Phormium tenax*	*Selliera radicans*
Phormium tenax		

Table 5.4. Plant occurrence and source at the Devil's Elbow bankworks.

Species planted	Species planted and survived	Species self-colonised
Juncus maritimus	*Juncus maritimus*	*Sarcocornia quinqueflora*
Leptocarpus similis	*Leptocarpus similis*	*Suaeda novae-zealandiae*
Schoenoplectus pungens	*Carex litorosa*	
Plagianthus divaricatus	*Plagianthus divaricatus*	
Carex litorosa	*Coprosma robusta*	
Coprosma robusta	*Phormium tenax*	
Phormium tenax		

Plant Growth

Where species are planted in two different tidal zones within a site, there is no significant difference in plant height between the tidal zones. For example, there is no significant difference in height between *Leptocarpus similis* growing at low and mid tide zones at Oxidation Pond Nos. 5 and 6 (Tables 5.5 and 5.6) and at the Devil's Elbow bankworks (Table 5.8). *Sarcocornia quinqueflora* shows the same trend at the Charlesworth Street Reserve (Table 5.3) and the Devil's Elbow bankworks (Table 5.7). Of these two species the between-site difference is only significant for *S. quinqueflora,* which is significantly taller at the Devil's Elbow bank works (Table 5.8).. Even though most species' heights do not differ significantly between sites, if the time of planting is considered some plants appear to have grown quicker at some sites. For example, *Leptocarpus similis* was planted at Oxidation Pond No. 6 in 1997 and has attained a greater height here (Table 5.6) than at Oxidation Pond No. 5 (Table 5.5) and the Devil's Elbow bankworks (Table 5.8), where it was planted up to 2 years earlier.

Table 5.5. Maximum shoot length (± SD) of revegetation species at Oxidation Pond No. 5.

Species	Low tide	Mid tide	High tide
Leptocarpus similis	33.31 (12.46)	37.58 (7.46)	-
Scirpoides nodosa	-	-	69.98 (20.33)
Carex litorosa	-	-	53.63 (12.17)

Table 5.6. Maximum shoot length (± SD) of revegetation species at Oxidation Pond No. 6.

Species	Low tide	Mid tide	High tide
Leptocarpus similis	44.03 (7.62)	43.75 (5.12)	-
Juncus maritimus	-	37.76 (9.41)	-
Scirpoides nodosa	-	-	47.69 (8.35)
Phormium tenax	-	-	84.42 (21.11)

Table 5.7. Maximum shoot length (± SD) of revegetation species at the Charlesworth Street Reserve.

Species	Low tide	Mid tide	High tide
Sarcocornia quinqueflora	6.9 (0.84)	10.61 (3.53)	-
Spergularia media	5.6 (2.26)	-	-
Selliera radicans	-	2.55 (0.64)	-
Juncus maritimus	-	52.09 (8.18)	-
Plagianthus divaricatus	-	-	57.17 (8.44)
Phormium tenax	-	-	89.96 (24.20)

Table 5.8. Maximum shoot length (± SD) of revegetation species at the Devil's Elbow bankworks.

Species	Low tide	Mid tide	High tide
Sarcocornia quinqueflora	-	16.13 (3.17)	15.79 (3.82)
Suaeda novae-zealandiae	-	-	25.00 (3.82)
Leptocarpus similis	39.00 (6.36)	42.58 (10.47)	-
Carex litorosa	-	-	60.80 (21.58)
Juncus maritimus	52.77 (10.66)	-	-
Plagianthus divaricatus	-	-	31.25 (12.93)
Coprosma robusta	-	-	52.25 (3.95)
Phormium tenax	-	-	82.00 (7.79)

Plant Condition Assessment

Condition score (averaged across species) was lowest at Oxidation Pond No. 5 and this site is clearly less successful than the other three projects (Fig. 5.10). Plant condition appears more variable between species (Fig. 5.11) than between sites (Fig. 5.10). This tends to indicate that the poor performance of the Oxidation Pond No. 5 site could be a site x species confounding problem (e.g. Oxidation Pond No. 5 has mainly poorly performing species). However, the species scoring poorly at this site are all recorded as scoring highly at other sites, indicating that it is the site rather than the species used causing poor condition. Site condition is indicated by Figs. 5.12-15.

Table 5.9. Mean rank (± SD) of revegetation species at Oxidation Pond No. 5.

Species	Low tide	Mid tide	High tide
Leptocarpus similis	1.71 (0.47)	1.81 (0.40)	-
Scirpoides nodosa	-	-	4.35 (1.30)
Carex litorosa	-	-	2.50 (0.55)

Table 5.10. Mean rank (± SD) of revegetation species at Oxidation Pond No. 6.

Species	Low tide	Mid tide	High tide
Leptocarpus similis	3.60 (20.50)	4.00	-
Juncus maritimus	-	3.90 (1.44)	-
Scirpoides nodosa	-	-	4.14 (0.36)
Phormium tenax	-	-	3.80 (1.10)

Table 5.11. Mean rank (± SD) of revegetation species at the Charlesworth Street Reserve.

Species	Low tide	Mid tide	High tide
Sarcocornia quinqueflora	3.67 (0.98)	2.67 (1.15)	-
Spergularia media	4.50 (0.58)	-	-
Selliera radicans	-	4.00 (1.55)	-
Juncus maritimus	-	2.00	-
Plagianthus divaricatus	-	-	4.63 (1.06)
Phormium tenax	-	-	4.75 (0.46)

Table 5.12. Mean rank (± SD) of revegetation species at the Devil's Elbow bankworks.

Species	Low tide	Mid tide	High tide
Sarcocornia quinqueflora	-	4.00	4.00 (1.03)
Suaeda novae-zealandiae	-	-	3.00
Leptocarpus similis	4.00	5.00	-
Carex litorosa	-	-	3.88 (1.55)
Juncus maritimus	2.57 (1.95)	-	-
Plagianthus divaricatus	-	-	5.00
Coprosma robusta	-	-	4.00
Phormium tenax	-	-	3.00

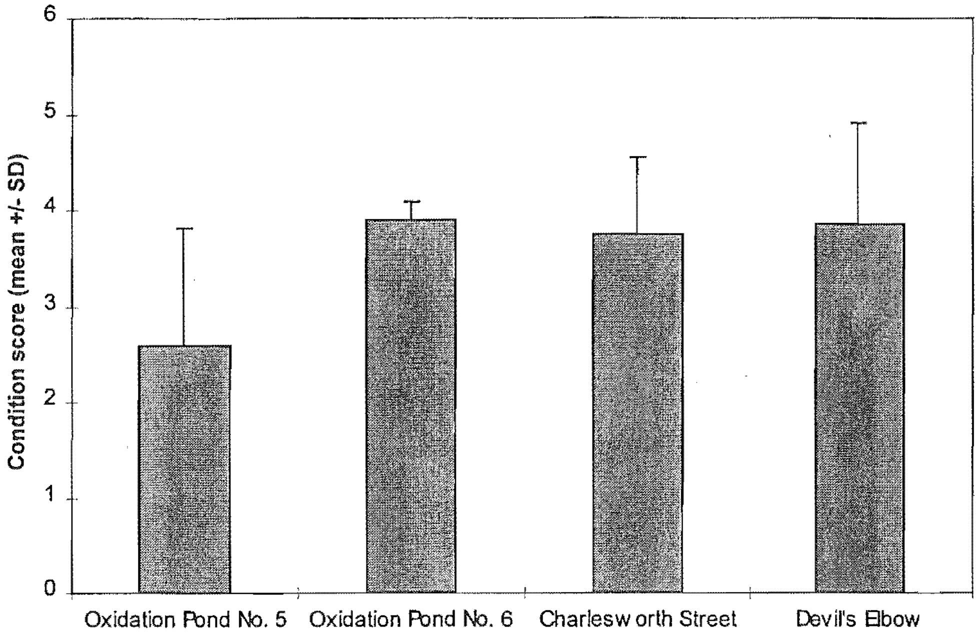


Figure 5.10. Condition score for each site, compiled from all species assessed at each site.

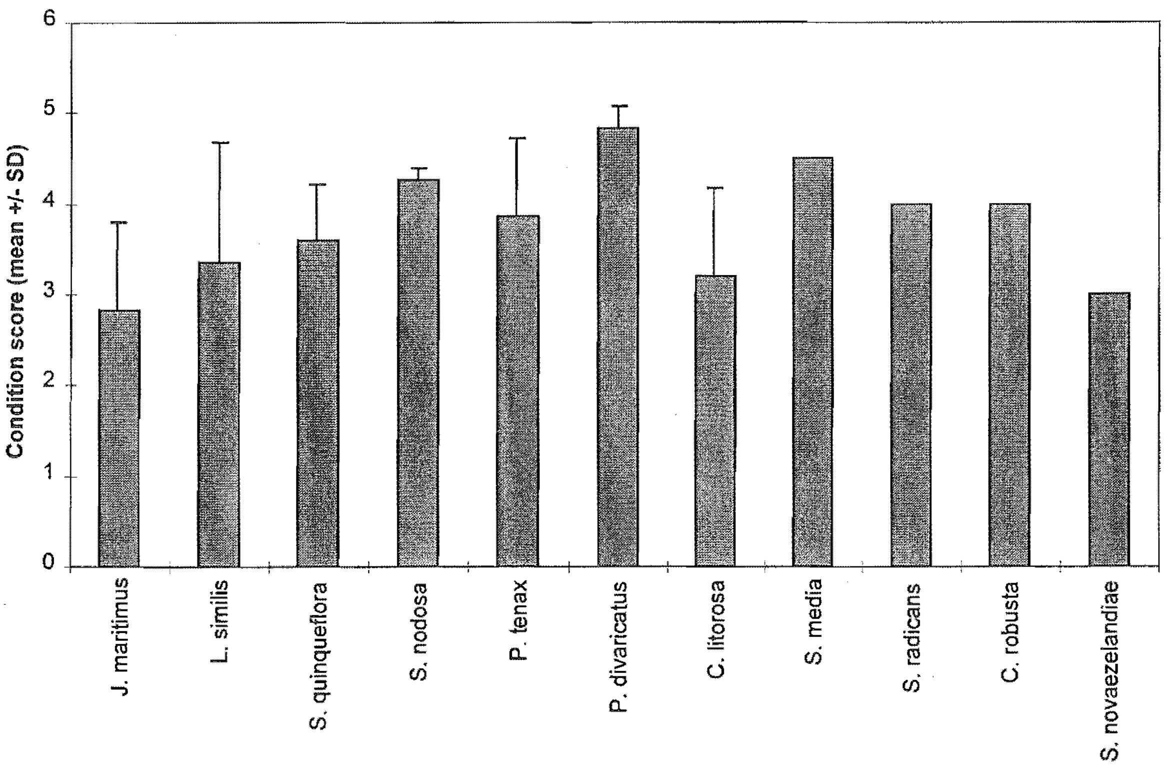


Figure 5.11. Condition score for each species compiled from all projects assessed.

Figure 5.12. Revegetation species along the margin of Oxidation Pond No. 5.

Figure 5.13. Revegetation species at the Devil's Elbow bankworks.

Figure 5.14. Pollution and a blocked culvert at the Charlesworth Street Reserve.

Figure 5.15. *Sarcocornia quinqueflora* at the Charlesworth Street Reserve.

5.4 Discussion

This study highlights the difficulty in learning from projects where there is a lack of detailed baseline information, design plans have not been followed accurately (Christchurch City Council, pers comm.) and subsequent management and monitoring have been lacking. If detailed planting records are not kept, species success and deciding which species have self-colonised are difficult to determine. It is unfortunate that despite the time and expense spent on designing and planting such areas, all that can be measured and used for comparison is a subjective assessment of plant condition. Adequate record-keeping and monitoring is essential if we are to learn from such projects.

Plant Occurrence

Salt marsh herbs rather than rushes or sedges appear to be the best species at self-colonising restoration sites with favourable substrates (Tables 5.3 and 5.4). Many of these species are succulents and their flowers are located in the joints between the stems (e.g. *Sarcocornia quinqueflora*). Therefore, they can be dispersed by the tide when fragments break off and lodge in suitable substrate. Once they have reached a site, salt marsh herbs spread vegetatively forming dense mats. *Juncus maritimus*, *Leptocarpus similis*, *Scirpoides nodosa* and *Phormium tenax* all appear to survive relatively well once planted. The failure of *Schoenoplectus pungens* once planted at all sites is consistent with that found in the transplant experiment (Chapter 4) and transplant trials on *S. pungens* by Partridge and Wilson (1988a).

Plant Growth and Condition

Generally plant condition was not high across the four sites. In particular plantings on the margin of Oxidation Pond No.5 were in significantly poorer condition than those found at the other sites. This is the site where a very coarse gravel substrate was being trialed. The coarse substrate appears to be the most obvious difference between this project, the other projects and natural marsh systems (Chapter 3). Such a coarse substrate does not appear to retain sufficient moisture and nutrients for adequate plant growth or colonisation.

The other obvious impediment to plant success is lack of sufficient tidal influence. Compared to the Charlesworth Street Reserve, the two Oxidation Pond sites and the Devil's Elbow bankworks have the advantage of continued exposure to unimpeded tidal flows. The Charlesworth Street Reserve relies on a single culvert to provide tidal flushing, meaning that post-construction maintenance is vital. At both assessment times the culvert has been blocked, indicating that maintenance is lacking. Lack of tidal flushing may explain the dominance of *Sarcocornia quinqueflora*, one of the most salt-tolerant species. Given that this was the only project specifically designed to facilitate salt marsh development, one would expect it to be far more successful than the other projects. Inadequate hydrology appears to be the reason for the poor plant performance.

Restoration Implications

The first requirement for successful restoration is a detailed plan which includes the following (adapted from Kusler and Kentula, 1990):

- clear project goals and measures for determining project success.
- boundaries of the proposed restoration or creation area.
- sources of water supply and connection to existing waters.
- proposed soils and probable sedimentation characteristics.
- proposed plant materials.
- indication of whether exotics are, or may be, present and, if so, what is to be done to control them.
- methods and timing for plantings.
- a monitoring program.
- proposed mid-course correction and project management capability.

Christchurch City Council designs for the four sites included all of the above to some extent, except they lacked a monitoring program, proposed mid-course corrections, maintenance (e.g. culvert clearance) and clear measures for success. Further research on the correct substrate is also required so that restoration attempts mimic the natural situation more closely.

It appears that if the correct substrate and hydrology are provided then salt marsh herbs (e.g. *Sarcocornia quinqueflora, Selliera radicans* and *Spergularia media*) will colonise on their own, making the planting of these species unnecessary if weedy species are prevented in Linwood Paddock restoration. Rushes (e.g. *Juncus maritimus* and *Leptocarpus similis*), sedges (e.g. *Carex litorosa*) and margin shrubs (e.g. *Plagianthus divaricatus*) would not be expected to self-colonise and will have to be planted where desired.

Although restoration projects should be designed to enable self-sustaining systems and persistent wetland features in the landscape, monitoring and mid-course corrections are still needed. The benefits of a good restoration design will only accrue with careful implementation and follow-up management. Such corrections relevant to these projects include replanting and alterations in hydrology. Some of these can also be continued as long-term management strategies (e.g. water level manipulation and herbivore control).

Given the high cost of demonstration projects, the greatest potential for filling gaps in scientific knowledge may lie with careful monitoring of selected types of new restoration or creation projects (Kusler and Kentula, 1990). It would certainly be advantageous to the success of future projects (including Linwood Paddock salt marsh restoration) if the Council monitored present attempts.

6. WASTEWATER TREATMENT AND THE POTENTIAL FOR PHYTOREMEDIATION

6.1 Introduction

Wastewater disposal is fast becoming one of Christchurch's most serious environmental issues. Current sewage effluent discharge into the Avon-Heathcote Estuary compromises the ecological value of this area and is unsustainable. Wetland restoration and creation can provide a multi-functional natural system that not only enhances wildlife habitat, but also purifies sewage effluent (Gearheart and Higley, 1993; Lofgren, 1993; Kadlec and Knight, 1996). A long-term wetland solution that improves water quality is timely as wastewater discharge consents are under review and Christchurch's population continues to grow. All disposal solutions must comply with New Zealand legislation and the guiding principles of Agenda 21 and the Ramsar Convention, thus they should be sustainable and consistent with Maori values for the area.

Final stage oxidation ponds at Bromley are suitable for the creation of freshwater wetlands with the potential for wastewater remediation. Effluent that has flowed through created freshwater wetlands and the remaining Bromley Oxidation Pond system (subsequent to primary and secondary treatment) would also enhance the sustainability and diversity of the proposed salt marsh restoration in the adjacent Linwood Paddocks, by providing a reliable, continuous freshwater supply. The proposed combination of fresh and saltwater wetlands in series, should provide additional water purification and wildlife enhancement without compromising the function or efficiency of either system.

This chapter outlines the hazards associated with sewage effluent and why there are concerns about current wastewater discharge into the Avon-Heathcote Estuary and surrounding waters. It goes on to assess the Christchurch system, including a review of the latest proposals for wastewater treatment and discharge. The potential for wetland-based, wastewater remediation to provide both the necessary freshwater inflows for salt marsh restoration in the adjacent Linwood Paddocks (Chapter 7) and to enhance water quality values in the Avon-Heathcote Estuary are discussed. Recommendations are made in light of the current environmental situation and the multi-purpose potential of such a system.

6.2 The Christchurch Sewerage System

Towards the end of the 19th century the rapidly expanding city of Christchurch experienced continuous outbreaks of water-borne diseases caused by the proliferation of unsanitary sewage disposal methods (mostly consisting of outfalls into the Avon or Heathcote rivers) (Wilson, 1989). In response, the Christchurch Drainage Board commissioned a programme of works to connect the sewers to a large sedimentation tank (replaced by two septic tanks in 1905) at Bromley, which then discharged effluent into the Avon-Heathcote Estuary via percolation from a soakage drain network within the farmland and sand dunes (Wilson, 1989). In 1962 the Bromley site was remodeled forming the basic structure of the current Christchurch Wastewater Treatment Plant and associated Oxidation Ponds (Wilson, 1989). However, the Woolston Industrial area with its outfalls of tannery and gelatine wastes continued to discharge to the estuary until an industrial sewer from this area was connected to the treatment plant in 1972 (Wilson, 1989).

Protection of both public health and the environment is the fundamental purpose of wastewater treatment. Sewerage systems effectively isolate sewage from the community, thus affording some protection from infectious disease caused by the accumulation of pathogens in sewage. Use or disposal of sewage and sewage products should not increase the risk of disease from such pathogens (Department of Health, 1992). In addition sewage sludge from municipal treatment systems is likely to contain toxic heavy metals as well as organic and inorganic toxins. Any toxins present must be prevented from entering the food chain (Department of Health, 1992).

Sewage Legislation

Sewage treatment and disposal is controlled by the legislative obligations of the Health Act, 1956, the Building Act, 1991, and the Resource Management Act, 1991.

The Department of Health is charged under the Health Act to advise local authorities in matters relating to public health, and to prevent, limit and suppress infectious and other diseases. The Resource Management Act (1991) guides waste disposal in a number of ways. Firstly, all persons exercising functions and powers under the Act shall recognise and provide for the relationship of Maori and their culture and traditions with their ancestral lands, water, sites, waahi tapu, and other taonga (Section 6 (e)). Secondly, discharge of contaminants into the environment (including onto land) requires either a rule in a regional plan, a resource consent, or regulations (Section 15). Minimum restrictions on granting certain discharge permits are that: a discharge cannot directly enter water, or, indirectly enter water, if it produces an objectionable odour, or renders water unsuitable for consumption by farm animals, or causes any significant adverse effect on aquatic life (Section 107).

Sewage Hazards and Concerns

Currently, Christchurch's Bromley plant provides primary and secondary treatment and continues to discharge its wastewater into the Avon Heathcote Estuary after oxidation pond detention, as it has done for the last 116 years.

Bacteria present in sewage includes faecal coliforms, *Salmonella* spp., *Vibrio cholerae* spp., *Campylobacter* spp., *Escherichia coli* spp., *Legionella* spp., *Listeria* spp., *Shigella* spp., and *Yersina enterocolitica.* The majority of enteric pathogens die off very quickly outside the human gut. However, *Salmonella* spp. may survive 11 to 280 days in soil (Kowal, 1982).

Viruses found in sewage include enteroviruses (67 types including poliovirus, echovirus and coxsackievirus, which causes meningitis), rotavirus, hepatitis A virus, adenoviruses, and HIV (Department of Health, 1992). Generally viruses do not survive well in the environment and their numbers decrease rapidly (Scorber, 1976).

Protozoa found in sewage include *Entamoeba histolytica* cysts, *Crytosporidium* spp., *Balantidium coli* and *Giardia intestinalis* (Department of Health, 1992). Protozoa can survive for only hours on exposed vegetation, but may survive for days in the soil

(Liu, 1982). However, most would not survive a municipal sludge treatment process as at the Bromley Sewage Treatment Works where temperatures are between 35°C and 39°C for approximately 21 days (M. Bourke, pers. comm.).

A comprehensive microbiological study of digested sludges from the Bromley Oxidation ponds was carried out between October and December 1990. Results showed no detectable levels of rotavirus, *Shigella* spp. or *Campylobacter* spp. (Department of Health, 1992). However, eight different serotypes of *Salmonella* spp. were detected in 14 of the 20 samples, enteroviruses were found in half the samples and faecal coliform levels ranged from 686 to 31,500 per gram of dry weight of sludge (Department of Health, 1992). The Department of Health does not regard the upper levels recorded at Bromley as safe, and therefore sludge-amended Linwood Paddock soils need to be controlled in such a way and for sufficient time that most pathogens are destroyed before members of the public can come into contact with the soil. The infective dose for *Salmonella* spp. is about one million cells for adults, but as little as 100 cells for young children or elderly people (Department of Health, 1992). Limits proposed in the rules for the United States Environmental Protection Agency for sewage sludge applied to agricultural or non-agricultural land are not more than 3 cells of *Salmonella* spp. per gram of volatile suspended solids (Department of Health, 1992).

A further concern on the use of sludge (assuming that the issue of pathogenic bacteria and public health has been attended to) is the level of heavy metals. Heavy metals can accumulate in the food chain if they are applied via sewage sludge to agricultural land. The transfer of sludge constituents from the soil to vegetation is a function of soil pH, ion exchange capacity, applied sludge characteristics, the cumulative application rates, and the vegetation type (Page et al., 1987). Indeed, heavy metal availability to plants is greater in anaerobic soil conditions, which would be the case, if a wetland were created on the Linwood Paddock site. Metals that are both toxic and bioaccumulating (e.g. arsenic, cadmium, lead and mercury) present serious threats to the ehvironment (Department of Health, 1992). Enhanced concentrations in soil and vegetation may lead to increased uptake and bioaccumulation of these substances in grazing animals (e.g. cattle or birds) (Department of Health, 1992).

Despite continuous modification to increase capacity and reduce odour and other harmful effects since 1962, discharge consent under the Resource Management Act runs out in 2001, and this practice may no longer be acceptable. Consequently, there is a $30 m upgrade of capacity (including volumetric, biological, screening and filtering) and odour reduction underway. This is due for completion by 2006 and further improvements could still be necessary for the new resource consent, due by 2001. The treatment process implications of this and the future location of the discharge are serious questions faced by the Christchurch City Council.

Creation of a multi-functional wetland system that further purifies wastewater is one way for the Council to ensure that improvements meet new consent requirements. Although, biosolid application to the proposed Linwood Paddock restoration site ceased in 1995, the heavy metal and nutrient levels measured for this area (Chapter 2) are higher than those recorded in natural marsh areas (Chapter 3). The current recorded levels do not pose a threat to wetland restoration on the Linwood Paddocks where topsoil is likely to be retained. However, high influent water quality from the Bromley Oxidation Ponds is required to prevent heavy metals from increasing to levels where they inhibit wetland functioning. Highlighting the need for a pollutant-stripping freshwater wetland (of sufficient size to accommodate the volume produced by Christchurch's present and expanding population) within the Oxidation Ponds prior to salt marsh supply.

Currently the Christchurch City Council's Waste Management Unit is reviewing wastewater disposal, and has commissioned consultants (see below) to present several options.

6.3 Review of Issues and Options (Woodward Clyde Ltd.)

The City Council commissioned the consultancy company Woodward Clyde Ltd. to produce an "Issues and Options Report", which would investigate a range of alternatives and suggest any favoured methods for wastewater treatment and disposal. The consultants presented the following treatment technologies and discharge options (summarised by Andrew Nichols, Waste Investigation Officer, Christchurch City Council). A summary of possible solutions and costs for the discharge is presented in Table 6.1.

Treatment Technologies

- Disinfection. Ultra violet radiation is the preferred disinfection method and involves purifying treated wastewater over banks of powerful ultraviolet "black light" tubes.
- Nutrient Stripping. To reduce high levels of nitrogen and phosphorus, a Biological Nutrient Removal (BNR) Plant would be installed at the at the Bromley Treatment Works. This method is used by the Rotorua City Council to reduce nutrient levels and prevent weed growth in Lake Rotorua.
- Modified Ponds/Wetlands. The existing ponds would be heavily partitioned using gravel banks to ensure that the entire pond area is used to treat the wastewater. Effluent from the ponds would then be "polished" within surface-flow wetlands built on the Linwood Paddocks before a final discharge.

Discharge Options

- Irrigation of farmland. The wastewater flow from Christchurch could be utilised to irrigate up to 6000 ha of cropping land. However, the effluent would need to be highly treated with UV disinfection and the BNR removal of nitrogen to prevent contamination of groundwater reserves. Contamination of groundwater is a serious issue, difficult to remedy once achieved and should be avoided. Furthermore, the removal of nutrients seems a waste when irrigated crops require fertiliser anyway.
- Aquifer injection. This involves injecting highly treated wastewater directly into the shallow groundwater aquifers.

- Estuary discharge. All options proposed include a combination of the preceding treatment technologies. The recommended options should make the Estuary safe for swimming (most of the time) and preserve the Estuary as a habitat for fish. However, only those with BNR will prevent eutrophication.
- Ocean outfall. This involves pumping treated effluent 2 km out to sea in Pegasus Bay via a pipeline. UV disinfection may not be necessary if the modified pond/wetland treatment option is also chosen and waste water is discharged out to sea after having been through a wetland system. Nutrient removal should not be required at all. The dilution achieved by an outfall of 2 km in length is sufficient to allow for safe shellfish collection in the surf zone, even if the effluent plume was to be driven directly ashore. The consultants further believe that the ready dilution of the effluent plume means fish will not be harmed and as the pipeline will be buried for most of its length, there will be no navigational hazard posed by the structure. However, even 2 km out and dispersed, the net affect will still be an accumulation and some degree of eutrophication.

Table 6.1. Summary of possible solutions with costings for the discharge of Christchurch wastewater (from Nichols, 1998). BNR = biological nutrient removal. UV = ultra violet.

Option name	**Treatment technologies**	**Discharge option**	**Capital cost $**	**Running cost $**
Land 1	BNR + UV disinfection + ponds/wetland	Irrigation onto farmland	350m	7.46m
Land 2	BNR + UV disinfection + ponds/wetland	Injection into shallow aquifers	258m	6.22m
Estuary 1	BNR + UV disinfection + ponds/wetland	Estuary	128m	3.58m
Estuary 2	BNR + ponds/wetland	Estuary	119m	2.79m
Estuary 3	Existing plant + UV disinfection + ponds/wetland	Estuary	15.6m	0.75m
Estuary 4	Existing plant + ponds/wetland	Estuary	7.4m	86 000
Ocean 1	Existing plant + UV disinfection + ponds	2km ocean outfall	55.5m	0.274m
Ocean 2	Existing plant + ponds/wetland	2km ocean outfall	56.6m	0.218m
Ocean 3	Existing plant + UV disinfection	2km ocean outfall	56.7m	0.507m

*The existing plant is defined as the Christchurch Wastewater Treatment Plant after the completion of the current $30m capacity upgrade.

Evaluation of Woodward Clyde Report

My first major criticism of this report is the reliance on capital-intensive technological solutions. Although wetlands and land applications are discussed as options, they are neither recommended nor credited with their full potential or value. Secondly, dispersal is confused with reduction in quantity, resulting in complacency when there should be urgency. Thirdly, there is a lack of biological knowledge and basis, with little regard for real ecological value. Lastly, economic cost, not real value, or even real cost appears to be the primary criterion for the favoured solution. "Economic value" as it is presently calculated does not relate to real value any more than "economic cost" corresponds to real cost (Goldsmith et al., 1972). For example, consumer prices do not distinguish between the gadgets and luxuries that we do not need, and such essentials as unpolluted water, air and food, on which a high standard of living actually depends (Goldsmith et al., 1972). In fact, market value tends to place greater value on the former, and we usually take the latter for granted. Despite these concerns, valid institutional options (e.g. education and user-pays) are outlined, and indeed, all treatment and disposal options are undoubtedly preferable to the status quo.

Reliance on capital-intensive technological solutions

Reliance on capital-intensive technological solutions is unsustainable. With the advent of higher energy prices and increased labour costs, these systems have become significant expenses for the communities that operate them. To suppose that we can ensure the functioning of the ecosphere ourselves with the sole aid of technological devices, thereby dispensing with the elaborate set of self-regulating mechanisms that has taken thousands of millions of years to evolve (Goldsmith et al., 1972), is somewhat unrealistic. In the long-term we need a reduction in the amount of waste produced and an increase in recycling. The dollars spent on fertiliser each year is surely an indication that such nutrients are needed on the land from which they came.

Ultimately, both the Estuary and discharge options are an attempt to solve one problem by shifting it to another sphere - thereby creating an indefinitely bigger problem. Sewage or any pollution that is dealt with close to its source, will only ever pose a serious threat in a confined locality and such localities can be easily monitored. Sewage that is discharged into the ocean has the means to travel a greater distance and cause adversity over a much wider range of organisms and habitats. Although the effects are obvious, the remote cause or source is not. Such far-reaching effects are difficult to monitor and even harder to ameliorate. The extent to which we are simplifying ecosystems and destroying and polluting natural systems, so that we are forced to provide technological substitutes, is a real cost (Goldsmith et al. 1972).

Ideally, if a problem is created on land it should be dealt with on land. The use of a wetland represents a compromise between the ocean and land options and a move away from expensive technological solutions which, incidentally, will still be required due to our current production of toxic waste, at least until institutional policies and practices come into effect and reduce them at source. Systemic substitution is needed, whereby technological substitutes are replaced by natural or self-regulating ones (Goldsmith et al., 1972). One might argue that hasty or disordered change is highly disruptive and ultimately self-defeating, but unfortunately the timescale imposed on any proposal is much abbreviated by the dynamics of exponential population growth, resource depletion and pollution.

Confusion of dispersal with reduction in quantity resulting in pollutant loading

Ecological processes can be disrupted by introducing either substances that are foreign to them or "natural" substances in the wrong quantities (Goldsmith et al., 1972). Therefore pollution "control" by dispersal, is not a control at all, but a way of playing for limited time. Dispersal can only ever be a temporary expedient (Goldsmith et al., 1972). In Appendix B of the Woodward Clyde report it is stated that "most discharges require dilution of metal contaminants to reduce the concentrations to acceptable quality criteria". It does not matter how dilute; since metals accumulate, neither dilution or dispersion will ameliorate their long-term environmental effects.

The volume of sewage is directly proportional to population numbers and can only be stabilised or reduced by stabilising or reducing the population (Goldsmith et al., 1972). Unfortunately, as the report acknowledges, disposal as agricultural fertiliser is generally not feasible due to contamination by industrial wastes and high transportation costs. In Christchurch, industrial and domestic wastes are mixed, so contamination of potential fertilisers by heavy metals and other pollutants is high. This could be overcome by separating the two sewerage systems enabling those who produce pollutants to bear all purification costs. Such accountability would encourage waste reduction and recycling of waste onsite.

Lack of accurate ecological basis

Modified ponds aside, the consultants appear to have been unduly negative towards the wetlands option. Section 8.2.2 of the report states that "wetland creation would cause the loss of over-wintering sites and wet pastureland, resulting in possible losses in Pukeko and South Island Oystercatcher habitat and winter food supply, thus affecting the sustainability of the existing populations". Since wetland creation only involves the lower portion of the Linwood paddocks, adequate habitat will still exist. Furthermore, swampy environments are these birds' natural habitat - use of pastureland is only making do in a modified landscape. Essentially, the creation of new wetlands would enable small, self-sustaining populations of species such as Bittern, Kingfisher, Pied Stilt, Banded Dotterel and Pukeko. The report itself states that currently the continued occurrence of such species within the Estuary is probably only achieved through immigration of birds from other areas. Certainly, New Zealand as a whole is definitely not lacking in wet pastureland; however, it is lacking in marshlands.

Invalidity of the economic assertions

The report estimates the cost of the significantly modified ponds/wetlands at $15.4 m - not allowing for the capital cost of the land, nor for any lost revenue resulting from it no longer being available for farming. As the land is already Council owned, and unable to be sold due to the original purchase agreement with the previous owners, the capital cost would in fact be negligible. Furthermore, any loss in farming revenue would be offset by lowered long-term treatment costs, potential tourism revenue and a range of environmental and wildlife benefits that are outside the narrow economic value system. Standard economics provides only one justification for choosing one option over another i.e. whether a thing yields a monetary profit *to those who undertake it* or not. Note that the benefits do not accrue to society as a whole. Standard economics disregards a vast array of qualitative distinctions and is a narrow judgement which gives more weight to short term gains because in the long run we are all dead (Schumacher, 1974). This attitude is further highlighted by a comment in Section 6.3.18: "the marginal benefit from the new areas (wetlands and holding pond) will be low. This is an example of diminishing returns." Again, short-term economic returns are paramount; wildlife or even long-term economic gains are unaccounted for.

Valid institutional options

The report does, however, outline some valid institutional options that are important if the City Council is serious about solving the long-term problems posed by waste water disposal. These involve applying the principle of user-pays or polluter responsibility, and public education and information. For example the majority of household insinkerator users are unaware of the final destination of their dinner party left-overs once sluiced down the plug hole. If they thought their waste would end up in the waters of their seaside resort and result in additional rates, they may think twice about using such devices. Indeed, the report states that household insinkerators can have a significant effect on wastewater loads. One reference gives an increase of 25 % on 5-day biochemical oxygen demand (BOD_5) load and 33 % in suspended solids (SS) load for domestic properties that have insinkerators. *If* the cost of additional purification were placed on those with such devices and *if* the environmental impacts were widely known, then a reduction in their use would be expected. Thus a less polluted environment is achieved at a lower cost. Additionally, if the waste were to be composted, further benefits, including cost and energy savings, would be attained. This highlights both the environmental and economic benefits achievable through institutional options (education and user pays). Unfortunately, although education is definitely a long-term solution to reducing the problem, it may not be effective soon enough. Institutional policy could provide further immediate shifts in consumer behaviour by limiting consumer options so that only environmentally compatible items and services are available (e.g. soap instead of detergent). Thus educated or not, the product that has the least impact on the environment is used. However, there is a limit to such institutional control. Education is still needed to inform the public of links between certain actions and the negative environmental reactions (not always universally obvious). Although negative feedback can have a positive environmental effect, education is one of the only ways society can learn about the potential for negative feedback, thereby hastening the learning process and ensuring that all practices are sustainable.

Education and today's effective communication system does not allow any person to claim ignorance. Therefore, we are all accountable. Furthermore, whether the City Council enforces user pays or not, we all pay dearly for such all-pervasive pollution.

6.4 Wetland Opportunities

Wetlands appear to perform all of the biochemical transformations of wastewater constituents that take place in conventional wastewater treatment plants (Brinson and Westall, 1983), and the organic filter, part of many wetlands, represents a significant, albeit finite, nutrient sink (Bastian and Benforado, 1988). Indeed, one hectare of tidal wetland can do the job of US$123,000 worth of state of the art wastewater treatment (Bellamy, 1993). Physical and chemical reactions are thought to be the primary mechanisms for nutrient and trace metal removal, with vegetation uptake secondary (Sereico and Larneco, 1988). The properties of wetlands that contribute to wastewater renovation are detailed below (Bastian and Benforado, 1988):

- High plant productivity contributing to increased vegetation uptake of trace metals and nutrients through direct assimilation into their tissues.
- Large adsorptive areas of sediment and submerged vegetation providing an extensive surface area for microorganisms to transport pollutants (Nu Hoai et al., 1998) and for algal growths to reduce their concentration.
- Tendency of sediments to become anaerobic enhances the retention of many compounds (this along with aerated conditions in the water column and upper sediment layer, allows many processes to occur at the same time, including; formation of relatively insoluble phosphorus, metal complexes and metal

sulphides, and the removal or conversion of nitrogen through nitrification, denitrification, ammonification and volatilization).

Therefore, in comparison with conventional secondary and advanced wastewater treatment systems, constructed freshwater wetlands may have several advantages. They are able to remove wastewater contaminants effectively, thus reducing BOD, suspended solids, nutrients, trace organics and heavy metals with low costs of construction and maintenance, and low energy requirements (Nu Hoai et al., 1998). Bastian et al. (1989) indicated that under appropriate conditions, constructed wetlands have achieved high removal efficiencies (40 - 90 %) for BOD, suspended solids, nutrients, heavy metals, trace organic compounds and pathogens from municipal wastewater. Indeed, bacterial and viral die-off in wetlands is reported to be greatly enhanced over conventional wastewater treatment systems due to the development of a more complex microbiological community (Venus, 1987). Little is known about the dynamics of viral reduction, however, wetlands can be expected to provide a 2 log reduction in bacterial levels with even a two day retention period (Venus, 1987).

Unfortunately, as with land application systems, wetlands that are over-loaded are likely to exhibit inefficient constituent removal (Bastian and Benforado, 1988). There are several possible mechanisms for metal retention by sediment: adsorption on cation exchange sites, sedimentation, precipitation and complexation with soil or organic matter (Faulkner and Richardson, 1990). Each reaction may be limited by saturation (Nu Hoai et al., 1998). Clay minerals and organic matter, in addition to hydrous oxide materials, are always present in wetlands and all contribute adsorption sites (Nu Hoai et al., 1998). Under the reducing conditions common in wetland sediments, heavy metals rapidly precipitate as insoluble sulphides, whereas oxides and hydroxides will form under aerobic conditions (Nu Hoai et al., 1998). In addition, for volatile elements (e.g. mercury), loss to the atmosphere may be the dominant removal path.

Constructed wetlands suitable for use within New Zealand may be either surface or subsurface with respect to water flow. The choice is dependent on the area available the functions required and public needs. Council staff have had preliminary discussions with the Tangata Whenua with regard to the values associated with this area. In the past this area was an important mahinga kai site, today, with the disposal of human waste in this area, the Avon-Heathcote is no longer used as a source of food for Maori (Heremaia, 1995). The Tangata Whenua support the retention and restoration of wetland systems that restore the natural qualities of this area (Heremaia, 1995).

Surface Flow System

These systems typically consist of basins or channels, with some sort of subsurface barrier to prevent seepage, soil or another suitable medium to support the emergent vegetation, and water at a relatively shallow depth flowing though the unit. The shallow water depth, low flow velocity, and presence of the plant stalks and litter regulate water flow and, especially in long, narrow channels, minimise short circuiting (Environmental Protection Agency, 1988). As the majority of chemical transformations take place in the shallow aerobic surface layer, it is the specific surface area that determines the level of treatment obtained in surface flow wetlands.

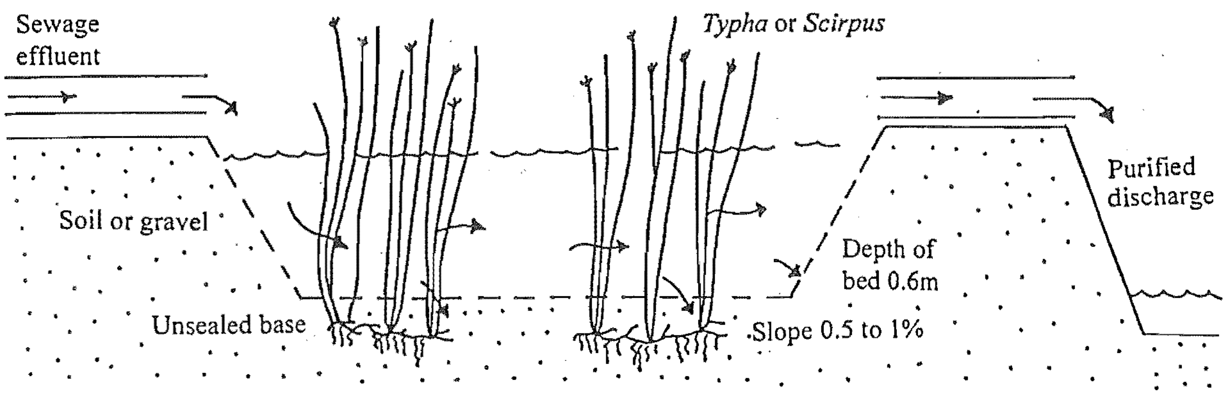


Figure 6.1. Emergent macrophyte treatment system with surface flow (adapted from Brix, 1993; Cooper, 1993). Purified discharge is released from the surface water layer of the wetland, once contaminants have settled in lower layers or have been chemically transformed.

As an example, in Brentwood and Dagenham, south-east England, to prevent hydraulic short-circuiting and maintain aesthetic appeal in constructed wetlands a series of weirs control the flow into three separate beds to treat metals from urban runoff (Scholes et al., 1998). In front of the first weir is a settlement zone for the initial removal of suspended solids. The first bed is planted with *Typha latifolia* followed by two beds planted with *Phragmites australis* - all planted at an initial shoot density of 4 m^{-2} (Scholes et al., 1998). The oxygen pump mechanism utilized by *Typha* and *Phragmites*, results in aerobic microsites adjacent to the plant roots in an otherwise anaerobic soil (Bastian and Benforado, 1988). A surface flow system was chosen because a sub-surface flow system would require a large cross-sectional area perpendicular to the flow and there was insufficient land available (Scholes et al. 1998).

In New Zealand, the Whangarei City Council and the Coromandel District Council have been successfully operating surface flow wetlands to polish sewage effluent for the past ten years. The Whangarei wetland covers and area of approximately 8 ha and is planted with *Scirpus*. It achieves a 50 % reduction in BOD and suspended solid removal but has little effect on faecal coliform levels (Duncan Thorpe, Whangarei City Council, pers. comm.). In addition, it is quite a local tourist attraction with over forty tours of the sewage plant and wetlands a year (Duncan Thorpe, Whangarei City Council, pers. comm.). Given that Whangarei's population is only 14 % of that recorded for Christchurch, to achieve similar removal efficiencies the size of the Christchurch wetland would have to be approximately 60 ha.

Subsurface Flow System

These systems are essentially horizontal trickling filters when they use rock media above an impervious base. They have the added component of emergent plants with extensive root systems within the media. Systems using sand or soil media are also used. Unlike surface flow systems, here the porosity of the medium is critical to predicting the required area for a given level of treatment (EPA, 1988).

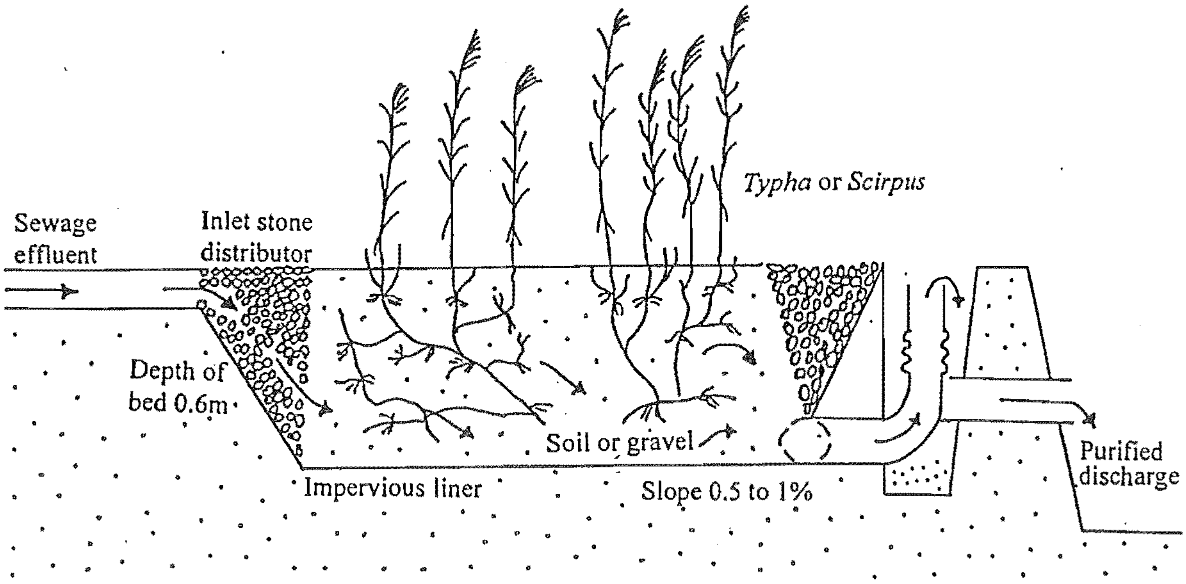


Figure 6.2. Emergent macrophyte treatment system with horizontal subsurface flow (adapted from Brix, 1993; Cooper, 1993). Effluent is aerated prior to discharge.

Subsurface reedbed treatment systems (RBTS) and constructed wetlands have been used in the UK since the turn of the century (Collinson et al., 1994). Thurlestone, a small coastal village in Devon and a popular tourist resort, has a good example of a subsurface reed bed (*Phragmites australis*) treatment system. Previously served only by an outfall that carried the village's crude sewage flow into the sea, the conversion to the wetland treatment system means that by the year 2001 a 250 % increase over original discharge quantities can be handled (Collinson et al., 1994). UV irradiation was considered, but would effect a reduction in only the bacteriological levels, and not the organic and inorganic content of the effluent (Collinson et al., 1994). Consequently, a subsurface RBTS was installed. Use was made of a gravel medium reed bed, thereby reducing the area required relative to a soil-based system. Iron-rich material and limestone chippings were also included in the medium, to maximise the potential for phosphate and nitrate reduction (Collinson et al., 1994). The scheme, as designed, met the main objective of providing consistent compliance with the EC

Directive limits for indicator microorganisms (Collinson et al., 1994). The constructed wetland has also developed and provided an extension to the natural habitat of the bay, both as a food source for wildlife and as a visually integral part of the local landscape (Collinson et al., 1994).

Plant Choice for Improved Water Quality

The remaining consideration in this section concerns the choice of freshwater, vascular macrophyte species, to be used in the final oxidation ponds (converted into freshwater wetland compartments). The species within these compartments will function to reduce effluent contaminants (suspended solids, N, P, bacteria, viruses, and heavy metals). Thus, "polishing" the remaining effluent after previous treatment. The next chapter will incorporate this initial and crucial freshwater wetland stage into a more comprehensive design that enables a salt marsh to be formed.

Macrophytes serve multiple functions in a wetland and are not just "attached-growth biological reactors," as many engineering-based reports indicate (Wetzel, 1993). They serve as major storage sites for carbon and nutrients, conduits of gases to and from the sediments, and regulators of water flow, allowing sediment deposition and toxicant retention. They also generate photosynthetically large amounts of organic carbon (Wetzel, 1993) and release nutrients in dissolved form that are readily available to other organisms.

A review of the overseas literature shows predominant use of the common reed (*Phragmites australis*) (Cooper, 1988; Scholes et al., 1998), raupo (*Typha spp.*) (Scholes et al., 1988; Shutes et al., 1993) and bulrush (*Scirpus spp.*) (Brix, 1993). Currently, there is no evidence that treatment performance is superior or different among these emergent wetland plant species (Kadlec and Knight, 1996). Indeed, metal analysis of plant tissue from *Typha latifolia* and *Phragmites australis* shows that both bioaccumulate trace metals (Scholes et al., 1998). At some sites *Typha* appears to accumulate more Zn, Pb, Cr, and Cd than *Phragmites*, whereas *Phragmites* tends to contain higher concentrations of Cu (Scholes et al., 1998), but concentration differences are not significant in any study.

A study by Venus (1987), for the Whangarei City Council, shows *Scirpus* to be more suitable than *Typha* for several reasons including: the reduced capacity for oxygen translocation to the root system, greater degree of litter fall which can cause problems with anaerobic conditions, and problems with windthrow if the roots are not adequately deep in the soil. Furthermore the use of *Typha* is associated with increased hydrogen sulphide production and odour, resulting from increased anaerobic conditions. *Scirpus* is also reported to encourage higher diversity of associated non-emergent species (Venus, 1987).

Both these genera have species native to New Zealand. Their use would pose no ecological threat. The use of native species and regional ecotypes poses no threat to biosecurity, adequate growth can be expected and habitat for native wildlife is enhanced. For example, pukeko (*Porphyrio melanotus*), New Zealand dabchick (*Podiceps rufopectus*) and the Australasian harrier (*Circus approximans*) utilise dead *Typha* for nesting, while during summer, *Typha* provides invaluable escape cover for unfledged duck and swan and for all moulting adult waterfowl (Ogden and Caithness, 1982).

Introduced species can be unpredictable in their effects and do little to enhance native ecosystems. *Phragmites australis* is another candidate for treatment wetlands but is not native to New Zealand. It is endowed with a vigorous rhizome and shoot system. Due to and greater height and density, it out-competes and excludes most other species (Colin Burrows, in press). It is also very fecund. Indeed, Holm et al. (1977) state that it is "capable of blocking up waterways and drainage channels". Such vigorous growth and associated nutrient stripping makes *Phragmites australis* ideal for use in a wastewater treatment wetland. However, with such attributes it poses a threat to biosecurity. It is classed as a *Surveillance Plant Pest* by the Canterbury Regional Council and is known to have caused serious adverse impacts in other regions (Colin Burrows, in press). Despite the obvious risks, use of *Phragmites australis* for nutrient stripping is being promoted by a North Canterbury company, Ocean Environmental Engineering, NZ Ltd. (Colin Burrows, in press). In addition, there has been an application for an exemption from Rule 7.1 in the Regional Pest Management Strategy (1998) to allow use of *Phragmites* in an engineered reed bed system for the treatment of sewage from the township of Southbridge, North Canterbury. The proposal is for the reed to be planted in an artificially constructed shallow basin, lined with impermeable plastic. The treatment plant will not be accessible by the public. Given the fact that *Phragmites* sets seeds, these precautions will not prevent the spread of this plant. Research into the use of a listed noxious weed is questionable, especially when native alternatives are readily available e.g. *Typha orientalis*, and *Scirpus* spp. Native alternatives should at least be researched in preference.

6.5 Freshwater Wetland Design for Wastewater Phytoremediation.

The primary purpose of freshwater wetland creation in this study is to purify Bromley wastewater, thereby enabling a regular freshwater supply to the proposed salt marsh restoration in the adjacent Linwood Paddocks. The Linwood Paddocks are already enriched with respect to heavy metal and nutrient levels (Chapter 2), therefore any freshwater supply should not contribute further to these already elevated levels. A freshwater supply (comparable to half that provided by the river flow of the Avon or Heathcote River) is necessary to sustain wildlife and functions in the proposed salt marsh.

If industrial pollutants were not so toxic and great in quantity, and if they could be separated from domestic wastes, I would recommend that the bulk of the sewage be used as nutrients on agricultural land and the residual water be run through surface flow freshwater wetlands (prior to discharge into the proposed salt marsh and ultimately the Avon-Heathcote Estuary). However, as I have outlined, the situation is not so simple and there will probably always remain pollutants of which we are unaware. Such pollutants make all discharge options hazardous to some extent. To reduce the pollutants associated with discharge, use of the existing plant makes economic sense and UV disinfection is required to reduce bacteriological levels. In combination with the initial ponds and created wetlands in the final ponds, the upgraded Bromley Treatment Plant should provide sufficient wastewater purification.

Indeed, Estuary or ocean discharge will be dependent on how efficient the treatment technologies prove. Given that 65 % of the discharged effluent mass currently remains within the Estuary, if the treatment technologies prove inefficient, Estuary discharge via a salt marsh is not a viable option. More distant dispersal into the ocean is not a long-term solution either. Thus reduction in wastes and the pollutant level in wastes is the only viable long-term option. However, the aforementioned examples indicate that freshwater wetlands alone are up to the task. In combination with the upgraded Bromley Treatment Plant and remaining Oxidation Ponds, they should provide ample purification.

A design for wetland wastewater remediation utilising the existing Bromley Oxidation Pond configuration is presented (Fig. 6.3). A surface-flow wetland system was chosen given the existing Oxidation Pond structure and the proposed multifunctional nature of the freshwater wetland. This system utilises the current pond set-up (without excessive modification) and also provides for wildlife needs (e.g. open water for waterfowl as well as vegetative cover and food). By redirecting Oxidation Pond water-flow so that Pond Nos. 3 and 5 are last in series (whilst still maintaining sufficient water retention time in the pond system), "polished" discharge can be directed to a high point in the adjacent salt marsh restoration (Fig. 6.3). This minimises premature mixing with tidal water and enables a gradient from fresh to salt water, thereby allowing for greater species diversity. There are various options for redirecting pond water flow prior to purification in surface flow freshwater wetlands (e.g. Woodward Clyde Ltd. present several options for Oxidation Pond modification involving the multiple partitioning of ponds). The important point, however, is that regardless of how the present Oxidation Ponds are modified, retention time in the Ponds must remain sufficient for water purification and that the final wastewater flow in the pond system is through created freshwater wetlands.

From analysis of other wetland remediation systems in New Zealand (e.g. Whangarei and Coromandel), Christchurch's current population of 339,500 dictates that the freshwater wetland would have to be at least 60 ha to achieve sufficient contaminant removal. Use of both Oxidation Pond Nos. 3 and 5 provides a combined wetland area of 91.5 ha. This area is sufficient to meet population demand now and in the foreseeable future. Furthermore, the use of two separate ponds allows for alternation in their use. Thus, the system will remain functional if maintenance is required on either one.

The planting design of the converted ponds could be analogous to that used in Arcata, California. The Arcata system utilises an intermediate treatment marsh system prior to final discharge into the constructed Arcata Marsh and Wildlife Sanctuary (EPA, 1988). Here the 'wetland ponds' are designed with several 15 m stretches of open space which span the full width of the marsh cell. The purpose of the open space is to provide a habitat for fish which will control the mosquito population and for wildlife (EPA, 1988). For this reason they used *Scirpus* as it allows fish better access to the planted areas (EPA, 1988). However, at Bromley the open space is more likely to continue to provide waterfowl habitat. If the wetland becomes overloaded, the channels can provide for sediment dredging to increase the efficiency of toxicant uptake and retention. Dredging of the initial freshwater wetland would mean that dredging is unnecessary in the more sensitive salt marsh and wildlife areas that follow. Furthermore, plant harvesting may be necessary to maintain open water. If dredged sediments and harvested plant matter are contaminated with heavy metals and other pollutants, a sustainable disposal method must be used and may require further research. Concentration of pollutants and relocation from area to another is not a sustainable solution to the problem. However, Venus (1987) states that research has shown harvesting to be unnecessary and even deleterious to effluent polishing and suspended solid retention.

A combination of *Typha* and *Scirpus* species would be most suitable for use in the planted wetland cells. *Typha* is renowned for phosphorus (Boyde and Hess, 1969) and heavy metal uptake and immobilization (Shutes et al. 1993). Thus, despite the drawbacks with *Typha* (outlined by Venus, 1987), I would still recommend its use in combination with *Scirpus*, due the inherent problems posed by a monoculture (disease susceptibility, overloading, toxicant removal specificity and limitations) and the wildlife benefits provided by *Typha*. The species should be planted in physically separated cells to minimise competition between the two.

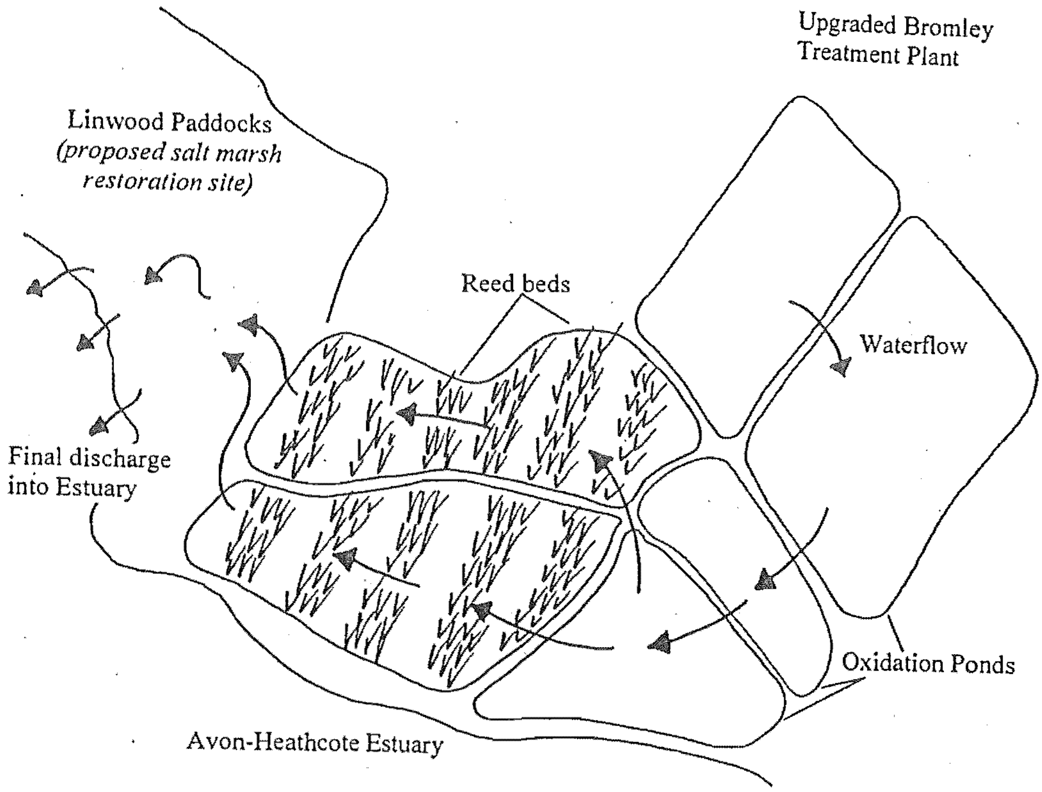


Figure 6.3. Freshwater wetland design for wastewater phytoremediation. Arrows indicate the necessary waterflow through the existing Bromley Oxidation Ponds prior to "polishing" in the final ponds that have been converted into wetlands through planting. Likely species for wetland reed beds include *Scirpus* and *Typha*. Multiple inlets and flow control berms ensure maximum utilisation of the wetland area and prevent short-circuiting.

6.6 Conclusion

Through conservative loading rates, high influent water quality, and simple mechanical controls requiring low maintenance (Kadlec and Knight, 1996), the recommended wetland treatment systems should provide a dependable freshwater source, resulting in high wildlife values in the restored salt marsh. Furthermore, it should meet final effluent limits as determined by the New Zealand Coastal Policy and the Regional Environment Plan indefinitely.

However, there are concerns about wetland 'aging' and decreased removal rates (Young, 1996). Little research has been conducted in New Zealand on emergent wetland plant species differences in relation to phytoremediation (Venus, 1987) or features of native macrophytes that could be utilized to maximise retentive or processing functions over the long-term. Further research could include growth and production experiments and harvesting studies. For example some researchers (e.g. Wetzel, 1993) believe that the macrophytes should be kept in r-growth stages by intentional, programmed disturbances to maximise contaminant uptake.

The use of wetlands for effluent polishing appears to have the general support of the local Tangata Whenua as the use of wetlands is seen to be more consistent with Maori aspirations in respect or avoiding direct discharge of effluent into receiving waters (Venus, 1987).

The potential for a variety of wetland systems to renovate or reuse treated effluent has been well established (Hammer and Bastain, 1990; Brix, 1993; Cooper, 1993). What these systems give up in terms of land requirements, susceptibility to environmental influences and lack of direct operational control, they make up through potential savings in energy, manpower, resources, and management costs and benefits (Bastian and Benforado, 1988). Certainly, the possibility for linking freshwater wetland phytoremediation with salt marsh restoration, thereby creating a multi-purpose wetland system is a win-win solution for Christchurch.

7. SALT MARSH RESTORATION: DESIGN, IMPLEMENTATION AND MANAGEMENT.

7.1 Restoration Goals

Restoration design for the Linwood Paddocks should be practical, though scientifically based and ultimately self-sustaining. The overall goals of the design are to re-establish a salt marsh wetland complex which is (i) representative of Canterbury salt marshes (ii) self-sustaining and (iii) accessible to the general public.

7.2 Restoration Objectives

To achieve the restoration goals six specific design and management objectives must be met:

(i) Establishment of salt marsh vegetation comparable in species composition and cover to that found "naturally" in Canterbury marshes.

(ii) Establishment of suitable habitat for native marsh birds, fish and invertebrates.

(iii) Establishment of links between the restored marsh, existing salt marsh remnants and the Avon-Heathcote Estuary.

(iv) Increased shoreline stabilisation and protection from predicted sea-level rise.

(v) Provision of additional water purification functions (Chapter 6).

(vi) Provision of a unique tourist and educational facility.

7.3 Restoration Strategy

The following recommendations for wetland redevelopment at Linwood are based on my research of Canterbury wetland pattern and processes, salt marsh species, the Linwood Paddock site, previous local revegetation attempts and literature research on wetland restoration in New Zealand and overseas.

Assuming sufficient social and political will, resources and a coastal site, there are three fundamental requirements necessary to achieve the restoration goals and objectives. Firstly, there must be a detailed understanding of local wetland structure and how it relates to function. Secondly, this understanding must be applied in the implementation of salt marsh features. Lastly, the system must be given sufficient time to become established.

Therefore, in the first instance, the Linwood system should be designed for function, not form. If initial plantings fail but the overall function of the wetland, based on the fulfilment of initial objectives, is being carried out, then the wetland has not failed (Mitsch and Cronk, 1992). To enable a certain amount of self-design (allowing natural recolonisation and establishment patterns) and to sustain restoration, hydrological links that supply wildlife recruits and nutrients should be maintained and enhanced. Tidal circulation and freshwater inputs can transport great quantities of water and nutrients in relatively short periods, supplying wetlands open to these flows (Mitsch and Cronk, 1992). Plant (e.g. seeds and stem fragments) and invertebrate recruits can also self-introduce from source wetlands by these flows. Therefore, hydrological links with the existing Charlesworth and Lovetts freshwater drains should be maintained, the tidal link between the Avon-Heathcote Estuary reestablished, and additional links with the Bromley Oxidation Ponds and freshwater wetlands (Chapter 6) created. This should ensure that the system is compatible with the local hydrologic landscape and climate, allowing for sea level rise, droughts and storms. Lastly, several years may pass before plant establishment, nutrient retention, and wildlife enhancement reach a level of sustainability - wetlands do not become functional overnight (Mitsch and Cronk, 1992). For example, the restoration of certain coastal salt marshes has been estimated to require at least 50 years (Frenkel and Morlan, 1991). Strategies that try to short-circuit or over-manage ecological succession are doomed to failure (Mitsch and Cronk, 1992) because knowledge of future environmental conditions, which have to be accommodated by the 'new' marsh will not be accurate enough to allow for them in advance.

7.4 Restoration Methodology

There are two general approaches for introducing wetland species in restoration projects. One is the "designer" approach of introducing species in a predetermined pattern or sequence and expecting their survival as planted or introduced (Mitsch and Wilson, 1996). The other emphasises the "self-design" capacity of nature both to recruit species on its own and to make choices from those species introduced by humans (Mitsch and Gosselink, 1993). To ensure efficient use of resources, whilst still providing for long-term sustainability, buffer and initial salt marsh species should be introduced first using the "designer" approach. Further salt marsh plant, animal and invertebrate species' introductions would be facilitated using the "self-design" approach. Following these principles, restoration requires initial site preparation that produces conditions analogous to that found in natural wetlands, planting and establishment of initial wetland plant species, allowance for natural modification of initial plantings, and facilitation of self-colonisation by additional wetland plant and animal species. If restoration is governed by environmental filters which select species from a pool of local biodiversity, the resulting system will be better adapted to the prevailing environmental conditions (Genet, 1997). After the initial period of revegetation and colonisation is underway, management is needed to maintain hydrological links and prevent any factors that may cause functional failure (e.g. high herbivory or dominance by weed species).

7.5 Restoration Site Preparation

Salt marsh communities frequently establish by themselves once the appropriate conditions are present (Knox, 1992). To facilitate the self-establishment tendency the paddocks would need to be lowered to the species' elevation ranges provided by the revegetation template in Chapter 3. Fortunately, the already low-lying nature of the paddocks means that if the tide gates (Fig. 7.2) are opened, unimpeded mean high tidal flows are predicted to reach over halfway (10.15 m above O.D.) into the paddocks at the current elevations (John Walters, pers. comm.) (Figure 7.1). Therefore, at certain locations site contouring need not involve moving large amounts of soil to enable the correct hydrology. At sites where earthworks are necessary (to lower elevated areas) the topsoil should be retained and returned as a covering for the newly created elevation. Salt marsh species have been shown to grow in Linwood Paddock topsoil (Chapter 4) and plants should be planted into this organic substrate, rather than into the lower subsurface horizons, which lack nutrients. Furthermore, early supply of organic soil also enhances diversity and species' persistence by containing the nutrients and structure necessary to support sufficient aquatic productivity. For example, a 12 ha lake in West Yorkshire, was created in 1987 as part of the restoration of an opencast coal mine site. Initially, waders were a

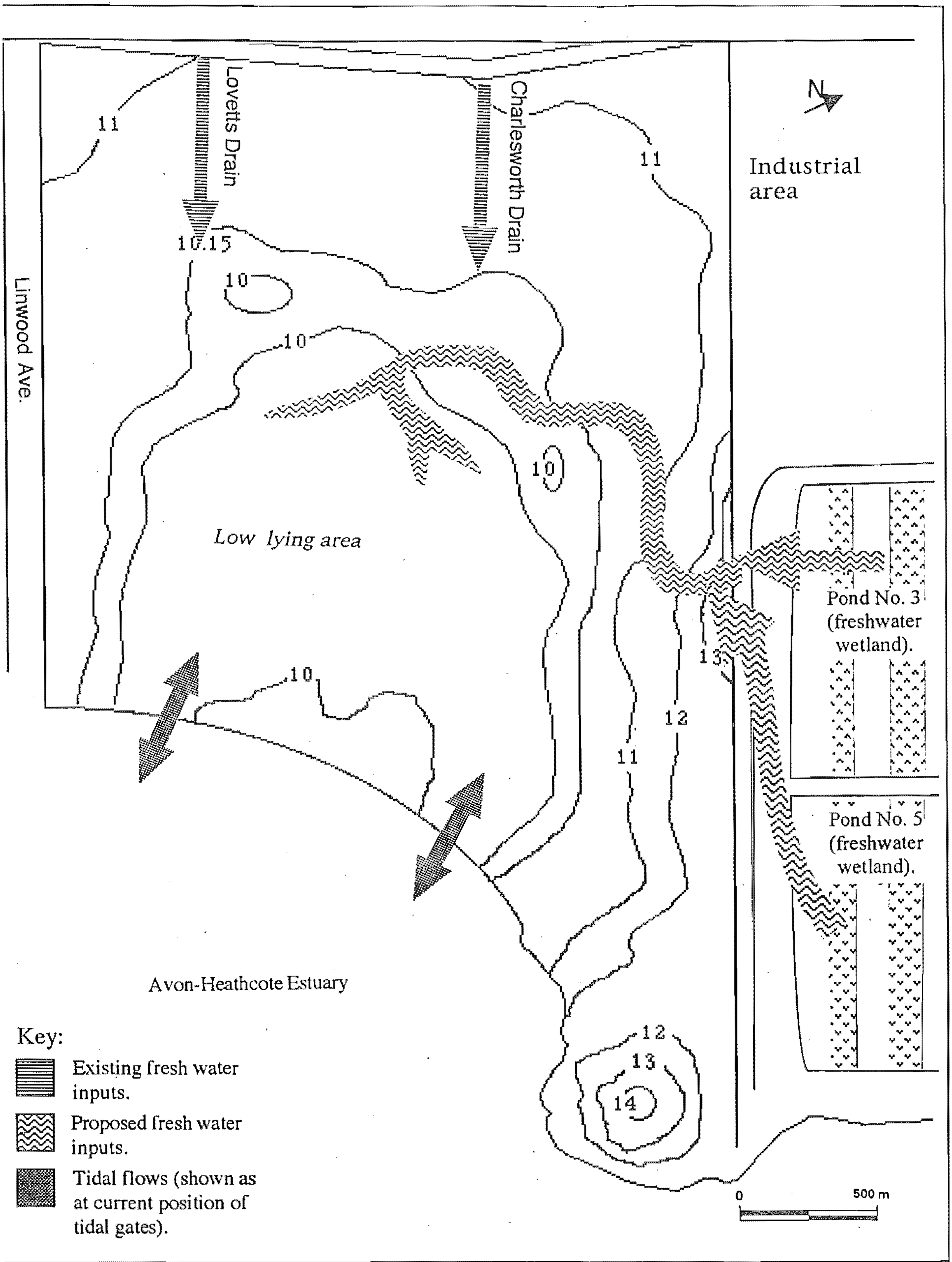


Figure 7.1. Topographical map of the Linwood Paddocks, indicating the proposed and existing hydrological links, which would enable a salinity gradient from salt to fresh water. Not only will these flows provide a water and nutrient supply, they will also facilitate species' recruitment and production export (once the marsh is established). Contour units are m above ordinance datum.

prominent feature of the lake's bird life and 29 species were recorded (Merritt, 1994). However, many of the migrant waders stayed for only a few hours or minutes, suggesting that although the lake was recognised as a potential food source, it did not offer them adequate food (Merritt, 1994). The clay substrate and minimal quantities of top-soil contained little or no organic matter, offering only low levels of plant nutrients and food for still water benthic invertebrates (Merritt, 1994).

To obtain a coastal salt marsh with a hydroperiod of semidiurnal flooding and dewatering superimposed on a twice-monthly pattern of spring and ebb tides, it will be necessary to either open the tide gates (Fig. 7.2) that separate the Paddocks from the Estuary, or remove the retaining wall (Fig, 7.4). To enable a gradient from fresh to salt water the Charlesworth (Fig. 7.3) and Lovetts Drains (with estimated daily low-flow discharges of 1045 m^3 and 275 m^3 respectively) must be maintained and additional freshwater (up to a maximum of ~ 135 000 m^3 per day) should be sourced from the created freshwater wetlands located in Bromley Oxidation Pond Nos. 3 and 5 (Chapter 6) by installing a culvert or a weir (Fig. 7.1). If only the tide-gates are re-opened and there is less marine input, then less freshwater is required to create a salinity gradient from fresh to salt water. To create the desired system (an analogous but smaller version of the Avon or Heathcote River marshes) the whole levee would have to be removed and more freshwater would be required. In this case freshwater inflow would have to be maintained at approximately 50 % of the Avon or Heathcote River flows.

Soil conditions on the site require monitoring prior to planting to ensure that salinity, moisture and pH are appropriate for the intended species (see Chapter 3), which are either planted directly or expected to self-colonise. Therefore, provision of adequate drainage is just as important as adequate flooding. Vagaries of excavation often leave pockets at lower elevations within which tidal waters stand after the tide recedes. Undrained areas are typically difficult to plant, and self-established seedlings do not survive, due to hypersalinity from evaporation and high water temperatures (Lewis, 1990). Therefore, all sites should be designed with a positive slope towards open tidal waters, and drainage channels should be placed at regular intervals to eliminate

Figure 7.2. Existing tide-gate between the Linwood Paddocks and the Avon-Heathcote Estuary.

Figure 7.3. The existing Charlesworth Drain and Linwood Paddocks.

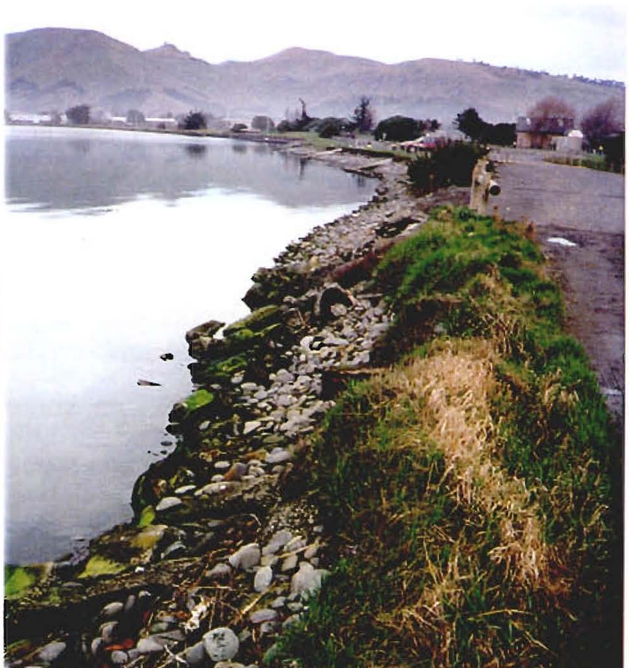


Figure 7.4. Retaining wall separating the Linwood Paddocks from the Avon-Heathcote Estuary.

stagnant pockets (Lewis, 1990). A system of channels that simulates natural creeks, not only allows for good tidal exchange and drainage, it also provides access to fauna (Broome, 1990). Greater use by fishes, benthos, and shorebirds have been reported where tidal channels are purposely created in man-made marshes (Newling and Landin, 1985).

7.6 Establishment of Initial Salt Marsh Vegetation

Native salt marsh vegetation is declining New Zealand. The number of native species is the most important of many factors that determine the conservation values of communities and thereby the restored areas (Knox, 1992). Even though salt marshes tend to be low diversity with respect to vegetation on a local basis (due to the monoculture tendency of marshes) on a regional or global wetland scale they can contribute to increased species diversity. For this reason all revegetation species suggested are native.

Buffer Species

Buffer species surrounding the restoration require planting first. The importance of adequate buffer zones around wetland developments is greatly under-appreciated. Buffer zones are essential for all reserved areas, as well as enabling one type of wetland development to co-exist with another (Environmental Council, 1983). At Linwood several functions can be performed by vegetation buffers at different locations. The northwest buffer should functionally imitate natural landward vegetation (especially that at the salt marsh/terrestrial interface) and then grade into suburbia (Fig. 7.6). This buffer must provide an effective screen for plants and animals between their normal or preferred habitat and disruptive human activities (e.g. traffic and urban expansion). It should also accommodate sea level rise. The northern and southern buffers should be narrower, and function to enhance wildlife connectivity between the Oxidation Ponds and the wetland (Fig. 7.6). Thus, encouraging wildlife utilisation. The eastern buffer (Fig. 7.6) is needed to provide shelter from the prevailing easterly wind.

In the first instance, buffer width can be determined depending on surrounding landuse and species sensitivity. If using vegetation alone, 30 m is generally recommended to reduce strong wind, noise and visual intrusion (Josselyn et al., 1990). This may be augmented by fencing to reduce human and pest (e.g. cats, rabbits and dogs) intrusion. However, sufficient fencing to prevent these pests would be a major and expensive undertaking. Furthermore, if this buffer is to accommodate sea level rise it may need to be wider. Artificial wetland boundaries often cut off the once in a 100 year extreme or the once in 50 year high water level of the wetland (Willard and Hiller, 1990). Less confined marsh boundaries enabled by a gently sloping buffer zone would allow plants and animals to adjust to changing hydro-regimes. Plants *in situ* must cope with extremes, but buffers enable populations to migrate, and adjust to, prevailing trends. Steep sides of confined systems remove the potential for adjustment and therefore force the loss of plant species, animals and habitat (Willard and Hiller, 1990). A wide-enough buffer zone would, therefore, enhance the quality of the salt marsh environment by allowing for environmental adjustment to predicted sea level rise. If the created salt marsh functions to increase sediment deposition at a rate that equals sea level rise, then the recommended 30 m may be sufficient. However, if deposition is predicted to be lower than the rate of sea level rise (likely in the early years) then the buffer must be wider (~ 60 m). Native species that can functionally imitate natural landward vegetation at the salt marsh terrestrial interface include; salt marsh ribbonwood *Plagianthus divaricatus*, mountain flax *Phormium cookianum*, and New Zealand flax *Phormium tenax*. Dr. Colin Meurk, Landcare Research, Lincoln (pers. comm.) and Knox, (1992) recommend use of the following species further inland to reduce urban disturbance, provide habitat and enhance aesthetic appeal (these species could also be used to enhance wildlife connectivity between the created wetland and the Oxidation Ponds):

- Species with a high tolerance to salt, wind and a good tolerance to wet soils: broadleaf *Griselinia littoralis*.
- Species with a high tolerance to salt, wind and wet soils: glossy karamu *Coprosma lucida*, karamu *Coprosma robusta*, cabbage tree *Cordyline australis*, *Leptospermum scoparium*, *Senecio huntii*, *Cortaderia* spp.
- Lowland coastal species once typical in Canterbury: ake-ake *Dodonaea viscosa,* karaka *Corynocarpus laevigata,* kowhai *Sophora microphylla,* kawakawa *Macropiper excelsum,* five-finger *Pseudopanax arboreum,* silver matipo

Pittosporum tenuifolium, putaputaweta *Carpodetus serratus* and mahoe *Melicytus ramiflorus.*

To provide shelter from the prevailing easterly wind taller trees must be sited at a right angle to the eroding wind. Furthermore, to avoid causing damaging turbulence it is important that the shelter filters the wind. This can be achieved by planting at pre-determined spacings (e.g. 3m in a 2 row design). Coastal buffer species must also have a high tolerance to salt, wind and wet soils. Cabbage tree *Cordyline australis*, ake ake *Dodonaea viscosa*, broadleaf *Griselinia littoralis*, ngaio *Myoporum laetum*, and akiraho *Oleria paniculata,* are all coastal species of reasonable height which could be used for this purpose. However, their roots are not tolerant to high salinities, and they must be planted above the tidal wash zone. The existing raised area close to Sandy Point may be a good place to plant the majority of these shelter species (Fig. 7.6).

Salt Marsh Species

Using the revegetation template in Chapter 3 (spatially represented in Fig. 7.6) as a guide, the salt marsh dominants which transplant well (Chapters 4 and 5), including *Juncus maritimus, Leptocarpus similis, Carex litorosa* and *Plagianthus divaricatus,* should be the first marsh species planted. *Sarcocornia quinqueflora, Selliera radicans, Spergularia media* and further salt marsh herbs, are expected to colonise on their own (Chapter 5) from source areas (Fig. 7.5). If native recruits fail to reach this 'new' marsh, they may be planted once the initial species have established. Additional wetland species that existed historically in Canterbury (e.g. *Baumea rubiginosa, Bolboschoenus caldwellii, Cyperus ustulatus, Eleocharis sphacelata, Lepidosperma australe* and *Sparganium subglobosum*; Colin Burrows, (pers. comm); could also be included at this less physiologically stressful later stage. Such an on-going initiative may be necessary to restore the full historical vegetation composition of Canterbury's wetlands.

Time of planting

Planting of high marsh and transition zone species must be done prior to the rainy season (e.g. March-April) unless irrigation is used (Josselyn et al., 1990). All other planting should be completed prior to the growing season, ideally in the spring (e.g.

September-October) (Shisler, 1990). Planting in summer may expose vegetation to extreme temperatures, salinity and dryness due to lack of tidal inundation (Shisler, 1990). Furthermore, planting in winter may expose plants to frosts.

Plant condition and start-up criteria

During the start-up period of constructed wetlands, lower water levels are desirable to avoid flooding emergent plants, accommodate a greater variety of vascular plants, and provide oxidation of organic sediments (Faulkner and Richardson, 1989; Mitsch and Gooselink, 1993). If transplanting from natural Canterbury marshes, Garbisch (1986) recommends a checkerboard removal technique to avoid the disruption of single large areas of wetland. It is essential to monitor the donor site to ensure that it is not degraded permanently by extraction of plant material (Josselyn et al., 1990). If transplanting is deemed too destructive and nursery stock is used instead, Garbisch (1990) recommends that plants should have been grown in a container long enough for root systems to have developed sufficiently to hold soil together. Furthermore, nursery stock to be transplanted to areas which are wet year-round should have been grown under hydric soil conditions for at least one growing season, thereby overcoming some of the growth differences observed between nursery and transplant stock (Chapter 4). Nursery stock must be grown from seed sourced from Canterbury marshes to ensure the ecotype is compatible with adjacent marshes and best suited to the Canterbury environment.

7.7 Encouraging Faunal Self-Introductions and Increased Species Diversity

Assuming adequate dispersal from source habitats (maintained primarily through hydrological links), animal species need not be physically introduced into the salt marsh. The habitat provided by the site conditions and revegetation should facilitate and encourage use of the salt marsh by a wide range of native species. Basically, if the salt marsh plants (primary producers) are provided for and their production and diversity are maintained, then the marsh fauna will be provided for.

Artwork: Don Rae — Inventas Media

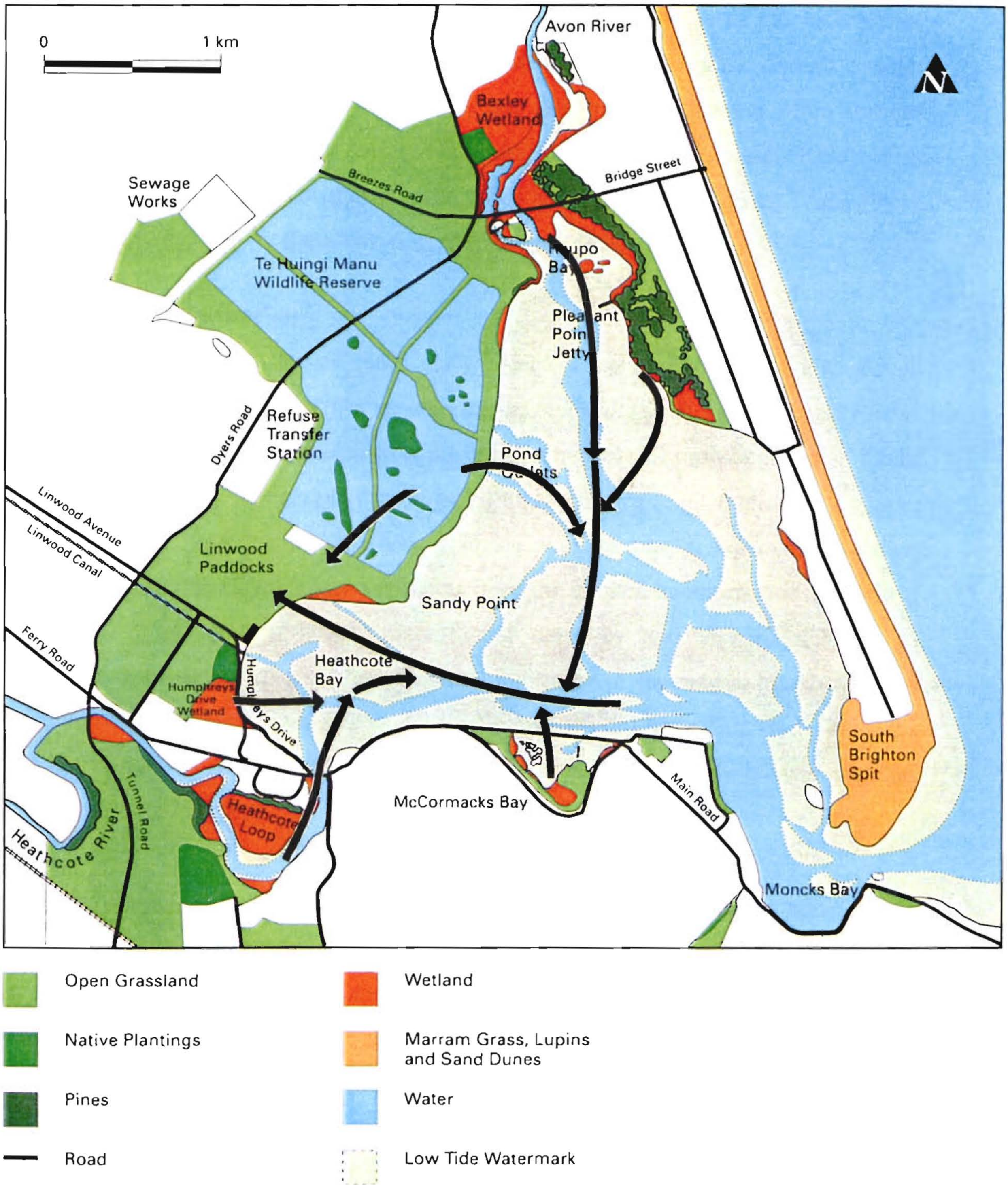


Figure 7.5. Arrows indicate the likely sources of recruits dispersed by surface water flows (e.g. seeds, plant fragments and invertebrates) for the newly restored marsh from salt marsh areas close to the Linwood Paddocks. Once established, the arrows would become bidirectional as the marsh exports production and provides recruits to other areas. Adapted from Owen, 1992.

There are a variety of ways in which species can reach a newly created marsh without human intervention. Tidal flooding (Fig. 7.5) is one transport mechanism which expedites and facilitates introduction of less mobile fauna (Broome, 1990). Regular tidal flushing and water renewal flushes marsh areas and increases the invertebrate and fish populations. Therefore, flushed mudflats have a higher overall species richness than constrained ones. Microbes and soil invertebrates are often sourced from the roots and attached soil of field-dug transplants (Broome, 1990). This provides increased forage for more mobile birds and larger fish (Barnett et al., 1994), which usually migrate on their own accord (Broome, 1990).

Invertebrate Colonisation

Taxa with planktonic larvae and early successional polychaetes (e.g. *Nicon aestuariensis*, *Eumenia sp.*, *Glycera americana*, *Scloecolepides benhami* and *Pisone sp.*) (Webster, 1997) are expected to comprise the majority of initial colonists. Crabs (e.g. *Helice crassa*, *Hemigrapsus crenulatus*) and molluscs (e.g. *Amphibola crenata*, *Mactra ovata*, *Micrelenchus tenebrosus* and *Potomopyrgus estuarinus*) (Webster, 1997) would be the most common epifauna once the created marsh matures. Levin et al. (1996) found that although macrofaunal densities and species richness of sediments in lower marsh areas came to resemble those of the natural marsh within 6 months, species composition and faunal feeding modes did not. This study indicates that there may be significant functional differences between newly created marshes and more mature systems.

Some invertebrates require fine detritus on which to feed and coarser materials or vegetation to provide shelter. To support invertebrates, a covering deposit need not be very thick; most aquatic larvae, for example, live in the upper 100 mm of substrate (Merritt, 1994). This will be provided by the Linwood topsoil soil capping of ~300 mm depth provided as a growth medium for revegetation species. In addition, Netto and Lana (1997) showed that the composition and abundance of invertebrates varied with plant cover and was highest in vegetated areas compared to unvegetated areas. They found that below-ground and dead above-ground biomass, presented the highest correlation with invertebrate density. Their findings suggested that live plant material is primarily used as a refuge or physical support rather than a food source, which is more likely provided by dead above-ground material (Netto and Lana, 1997).

Highlighting the importance of retaining organic Linwood Paddock topsoil as a food source for initial invertebrates in the restored marsh.
In the established marsh, the maximum number of invertebrates would be expected to occur in late spring and early summer (Bella and Davis, 1995).

Marsh Bird Colonisation

A significant proportion of New Zealand's resident birds feed, moult, find refuge from winter, congregate for pairing and courtship, and raise their young, in wetlands. New Zealand wetlands support over 50 resident water bird species (Water and Soil Division, 1982). Many of these may require more than one wetland or wetland type during their life cycle and are dependent on a chain of suitable wetlands; therefore it is beneficial for wildlife if a regional variety of wetland types is retained (Water and Soil Division, 1982). The restored salt marsh would be part of an important local (Fig. 7.5) and regional link in the chain of wetlands along the central Canterbury coast between the Waipara River mouth in the north and the Rakaia River mouth in the south. Permanently (all season) marsh birds include bitterns, crakes, rails and fernbirds. These birds dwell almost entirely in dense thickets of aquatic emergent vegetation, and in rush and shrub associations on waterlogged soils (Water and Soil Division, 1982). Wading and coastal birds will also likely use the marsh. However, they should not be catered for in preference to marsh birds, as sufficient habitat is provided for these species by the existing Avon-Heathcote Estuary and Bromley Oxidation Ponds (Te Huingi Manu Wildlife Reserve).

The Avon-Heathcote Estuary is part of the East-Asian flyway and is an important stopover site for thousands of birds. At peak times this wetland system supports a combined population of over 150,000 wetland birds (Crossland, 1992). The marsh may be either a destination or simply a transit stop for birds moving between the high country and the coast, from South Island to North Island, or even between the high Arctic and Australasia or the South-West Pacific (Crossland, 1992).

There is segregation of birds depending on habitat type. Unexpectedly, pied stilts have been found to favour the 'new' Charlesworth Street Reserve (Love, 1997). Godwits and oystercatchers were found in greatest numbers at Sandy Point and herons have been found in all marsh sites, at densities greater than in the Estuary proper

(Love, 1997). Analysis by Barnett et al. (1994) shows that wetland stopover preference is related to the sum of seasonal brackish and mudflat areas. The greater the acreage of these habitat types, the greater the usage by migratory birds. All of the wetlands designated as major Pacific Flyway wetlands have moderately high acreages of these habitat types (Barnett et al., 1994). Creation of a salt marsh adjacent to the estuary will increase the amount of shallow brackish water available to such migratory birds.

Fish Use

Twenty-eight species of fish have been recorded in the Avon-Heathcote Estuary and the Estuary fragments (Webb, 1972; Nairn, 1998). Eleven of these are common, mostly permanent residents, including sand flounder, *Rhombosolea pleia*; yellow-bellied flounder, *Rhombosolea leporia*; common sole, *Peltorhamphus novae-zealandiae*; yellow-eyed mullet, *Aldrichetta forsteri*; kahawai, *Arripis trutta*; lamprey, *Geotria australis*; spotty, *Pseudolabrus celidotus*; cockabully, *Tripterygion nigripenne*; giant bully, *Gobiomorphus goboides*; and common bully, *Gobiomorphus basalis*; and globe fish, *Spheriodes richei* (Webb, 1972; Eldon and Kelly, 1992). These species use a salt marsh in flood (an extension of the estuary) for adult and juvenile feeding. Indeed, Nairn (1998) found abundant juvenile flatfish and yellow-eyed mullet in channels close to all of the Avon-Heathcote marsh areas. Access to wetland vegetation for fish must be maintained by providing unvegetated channels throughout the wetland (Marble, 1992). Most of the species listed feed on crustaceans, molluscs, polychaetes, coelenterates and algae (Webb, 1973). All of these groups should be found in salt marsh sediments, provided productivity and hydrological links are maintained.

7.9 Provision of Shoreline Stabilisation and Sea-Level Rise Protection

Establishment of sufficient salt marsh vegetation cover over a typical (low-gradient) salt marsh profile would result in one of the least erosion-susceptible coastal environments and provide some protection against predicted sea level rise for this area (see Chapter 2).

A tidal wetland represents a natural balance between sedimentation due to flooding and erosion. It follows then that design of a suitable tidal wetland project must be able to evolve in response to natural episodic events, and adjust to long-term sedimentation (Goodwin, 1994). Natural coastal marshes dissipate energy associated with waves, currents and storm surges (Knutson, 1988). Certainly, frictional resistance is higher when water spreads out over a large area, rather than being confined to a channel. Therefore, to increase frictional resistance and decrease potential shoreline erosiveness, the wetland should be designed to allow sheet flow rather than purely channelised flow. However, flat systems are only stable in low-energy environments; higher erosion or deposition rates will change their profiles as seen in coastal dune systems. Therefore, a flat salt marsh structure will only provide short-term protection against storm surges. The key salt marsh effect is more likely the erosion-resistant roughness, provided by the salt marsh vegetation (e.g. *Juncus maritimus* and *Leptocarpus similis*). These plants are rigid, persistent, and tall enough to penetrate the entire water column. Thus they decrease water velocity by absorbing surge energy and increase sediment deposition, thus decreasing erosion. Under conditions of abundant sediment supply, the marsh could even prograde seaward (Knutson, 1988). In densely vegetated marshes, more than 50 % of the energy associated with waves can be dissipated within the first 2.5 m of salt marshes, and virtually no wave energy persists 30 m into such marshes (Knutson, 1988).

In addition, all established plant species would anchor the shoreline by binding the sediment with their root systems and measurably increasing sediment shear strength (Knutson, 1988). In studies summarised by Gray (1974), the shear strength of vegetated soils was two to three times greater than that of unvegetated soils. The importance of shear strength is clearly illustrated when one examines the process of erosion in established coastal marshes. Marshes are not typically eroded by the mobilisation of surface sediments because the marsh vegetation produces an energy dissipating environment (Knutson, 1988). In established marshes, erosion is frequently a more cataclysmic event, in which the marsh sediment fails (shears) in blocks on the margin (Knutson, 1988). The rate at which these blocks of marsh are carved from the marsh margin will be directly related to the shear strength of these marsh sediments (Knutson, 1988).

The global rise in sea level over the past 40 years has been estimated at about 3.0 mm a year (Emery, 1980) (see Chapter 2 for local predictions). This global (eustastic) rise is attributed to the melting of polar ice and thermal expansion of ocean waters due to increases in water temperature. However, the apparent rise in sea level is more variable if one considers changes in the elevation of coastal areas. Coastal subsidence is a common phenomenon in the United States due to ground water withdrawal (Knutson, 1988). For example, after withdrawal of subsurface fluids the Galveston Bay area is experiencing an apparent rise in sea level of about 4.0 mm/yr (Knutson, 1988). Fortunately, salt marsh accretion generally equals or exceeds local sea level rise (Nixon, 1980).

7.10 Provision for Education and Tourism

There are many education and tourism opportunities, both throughout salt marsh implementation and once established; it is a chance to capitalise not only on the fascination of people in general for interacting with animals and plants (Robinson, 1989), but also their desire to identify with the young, but rich, human history of New Zealand (Nature Conservation Council, 1981). However, the general public and tourist will only appreciate the full value of the ecosystem provided by salt marsh restoration if there is comprehensive interpretation. Miller et al. (1994) consider that without interpretation, New Zealand's heritage is like a full bottle without a label, valuable only to those who already know what it is. Therefore, a full range of field interpretation boards and a network of tracks and boardwalks (that extend out from access points and a visitor centre) should be planned before the beginning of planting. Boardwalks are advantageous in that they allow the public to get close to nature without impacting significantly on it (other than during initial construction disturbance). They also raise the viewer giving greater visual appreciation of the area. However, research has shown that the presence of boardwalks can affect benthic assemblages. In studies by Kelaher et al. (1998b) benthic macrofauna were affected within 3 m of the boardwalk resulting in fewer amphipods, gastropods and insect larvae than in areas up to 24 m away. However, a further study by Kelaher et al. (1998a) showed that boardwalks actually increased semaphore crab numbers. If constructed prior to any reestablishment of wildlife, boardwalk presence does not appear to pose a threat to wetland functioning.

The interpretation facilities should convey the conservation value of the salt marsh and its importance in terms of the estuary. Background information on the site and project purpose would also enhance the visitor experience. At entrance points information detailing restriction to walkways or boardwalks and prohibition of dogs or litter should also be provided. Information on the species likely to be present and the breeding pattern of those resident (i.e. times when extra caution is necessary) would also be advantageous. The breeding season lasts from July, when most birds return to their nesting territories until December (Crossland, 1992). From late November to early December onwards, birds too young to breed, and those that nested early, arrive at the estuary and associated wetlands to winter (Crossland, 1992). Therefore, the highest numbers of birds should be recorded between December and June.

In addition to information on natural, historic and cultural features, Miller et al. (1994) found that the public appreciates signs recording such things as the year of planting of particular species. The changing details of monitoring results would also add another interest point and education feature.

Certainly, one of the most powerful advocacy tools available to restorationists is the promotion of recreation in conservation areas (Booth, 1990). Many people simply want to experience the pleasure of seeing native species in a natural environment (Robinson, 1989; Nature Conservation Council, 1981). However, people in New Zealand have a growing awareness of, and conscience for, environmental issues, and increasing numbers of them want to experience conservation first hand (Miller et al., 1994). Opportunities to take part in activities such as planting or watching rare species being released are eagerly seized upon (Craig, 1990). Desire for this sort of experience can be used effectively at the Linwood Paddocks, where members of the public could combine recreation with a sense of achieving something of worth for Canterbury (Miller et al., 1994). By capitalising on this ethic, the Christchurch City Council can educate the public and enhance their conservation awareness, thereby also gaining public support (Booth, 1990).

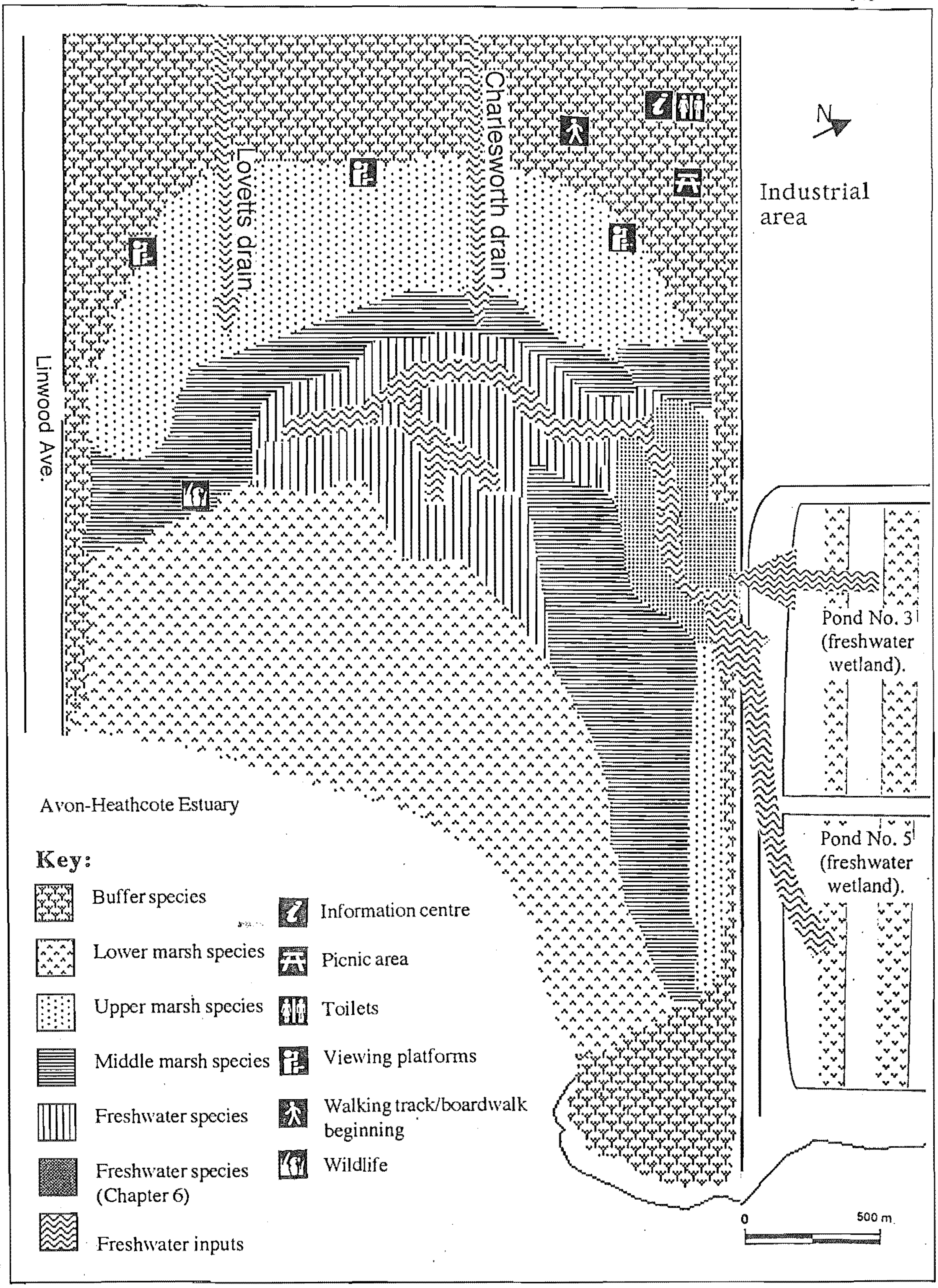


Figure 7.6. A spatial representation of the revegetation template (Chapter 3). Buffer and freshwater species (Chapter 6) should all be planted as "designed". The majority of upper marsh species would be expected to self-introduce as predicted by the revegetation template. Lower and middle marsh species would be a combination of species planted according to the revegetation template and self-introduced species. Signs indicate the potential for tourist and visitor use. Information kiosks and picnic sites will be located in areas of the buffer zone that are more sparsely planted (this should not cause gaps in the buffer if the outer-most buffer layer is densely planted). A 'stepped' entrance would allow public access whilst still providing a visual and wind barrier.

7.11 Management After Construction (minimisation of threats to wetland functional success)

In a less modified setting, restoration of the correct tidal regime and facilitation of revegetation would allow natural succession to proceed unhindered and a salt marsh to develop. However, in the more modified setting at Linwood, the results of minimal management may be undesirable. Thus, continued monitoring and adaptive management of the site is required. This need not be intensive and does not mean curtailing natural succession to achieve aesthetic goals, but rather to obtain (in a modified landscape) functions and attributes comparable to those in a more pristine setting.

Willard and Hiller (1990) state that management must aim for persistence, not constancy. Certainly, managed habitats can fail by becoming too constant and limiting their adaptability (Willard and Hiller, 1990). In this case the aim is for the wetland to support a persistent collection of wetland functions and not a wetland that looks a particular way. However, form and function may be related, since some wetland types do perform some functions better than other types (Willard and Hiller, 1990). Consequently, the manager/regulator has to find a combination of form and function which persists.

Specific Management Policies

(i) To enhance native plant growth and productivity.

(ii) To reduce wind and herbivore disturbance to marsh vegetation, and wind, predator, noise and other human disturbances to wildlife.

(iii) To enhance marsh and shoreline stability.

(iv) To enhance the nature conservation value of the created marsh and the estuary as a whole.

(v) To develop an improved understanding of salt marsh resources by the general public i.e. educational functions.

Monitoring Regime

Monitoring of the restored marsh is desirable to measure the project success and to determine additional needs or inputs such as replanting, control of undesirable plants, removal of litter, control of wildlife pests, and human traffic (Broome, 1990). If monitoring is to be an effective tool, the results must be redirected towards increased understanding of the functions achieved by the restoration project, and then used to effect changes which positively influence the desired functions and help to achieve restoration goals (Shisler and Charette, 1984).

Stratified random sampling should be used in each elevation zone, and should begin at the completion of construction and planting. Percent survival combined with growth and stem density measurements, provide a measurement of success and a good indication of whether the system is healthy and expanding in plant height and cover (Lewis, 1990). Results should be compared with parameters measured in "natural" Canterbury marshes i.e. percent cover (Chapter 3), growth and survival (Chapters 4 and 5). The first measurement may be an estimate of survival rate four to six weeks after transplanting. This provides a measure of initial planting success, including quality of the planting material, planting methods, and the effects of wave action (Broome, 1990).

The water chemistry should also be monitored, with salinity and pH levels maintained within those outlined in Chapter 3 for each species. The pH of Bromley Oxidation Pond water ranges from 7.2 – 8.8 (Mike Gilson, pers. comm.) and water sourced from here should be compatible with wetland functioning which is optimal at neutral pH 7. Dissolved oxygen concentrations greater than 4 mg/l or 60 % saturation are optimal and should be maintained at this level (Marble, 1992). Oxidation Pond water with a variable dissolved oxygen concentration varying from 0.3 – 19 mg/l and a median value of 7.5 mg/l (Mike Gilson, pers. comm.) would, therefore, be suitable for wetland diversion. Furthermore, although low levels of suspended solids less than 80 mg/l contained in runoff and surface waters entering a wetland have a high correlation with aquatic diversity and abundance, the maximum level should not exceed 200 mg/l (Marble, 1992). Suspended solid level in the final Oxidation Ponds is well under this maximum level; ranging from 15 – 100 mg/l with a median value of 50 mg/l (Mike Gilson, pers. comm.).

A photographic record is also an effective, low-cost method of monitoring a planting site, as well as providing a visual education tool that can be used in instruction for future projects or as a point of interest to visitors and tourists. Permanent stations should be marked at the site for photographs at each visit to record growth and development of the vegetation (Broome, 1990).

The typical standard sampling regime for determining success in tidal marsh systems is quarterly sampling for 2 years (Lewis, 1990). However, because of the stochastic nature of hydrologic events and the slow development of ecosystems due to the variable rate of recruitment and growth (Mitsch and Wilson, 1996), the 2 year time horizon can be viewed as arbitrary and probably much too short. Indeed, most reports state that marshes take at least 20 years to become fully functional and may take up to 50 years (Frenkel and Morlan, 1991). Therefore, in terms of measuring salt marsh success or lack thereof, after 20 years the earliest estimate could possibly be made. Short monitoring times only favour measuring success with transplanted vegetation and pioneer species; long-term success is less dependent on these initial conditions (Mitsch and Wilson, 1996). For education and maintenance purposes, monitoring could be continued indefinitely.

Minimisation of Threats to Wetland Success

Insufficient water supply and hydrological links

For salt marsh species' survival it is essential to maintain the correct hydrological regime. The most significant problems identified with created wetlands have been incorrect water levels and hydroperiod (Mitsch and Wilson, 1996). Culverts and other water-control structures from the Drains, Estuary and Oxidation Ponds must remain operational to maintain the gradient from fresh to salt water. It has been found that *Sarcornia* growth rates decrease by approximately 30 % where tides are excluded (Callaway et al, 1997). Indeed, this is the main reason for the limited success of most species at the Charlesworth St. Reserve (Chapter 5). Dredging of channels to maintain water flows may also be necessary if sedimentation is excessive. If drought conditions prevail, plant survival may be increased by additional watering (salt marsh plants do not require salt water for survival and will survive on freshwater). This may be provided by increasing the Bromley Oxidation Pond water input. Furthermore, salt marsh ecosystems can only positively influence other systems by production export, or provision of plant and animal stocks, if sufficient hydrological links are maintained. Certainly, a notable feature of wetlands with high species diversity is a permanently open ocean entrance (Barnett, et a., 1994).

Invasive plant control

The most commonly occurring weed species in this area are those that colonise disturbed sites (Heremaia, 1995). Unfortunately, the largest numbers of these species occur along the Avon-Heathcote Estuary margins that will be linked directly to the restoration site. Problem species include; spartina *Spartina anglica*, tall fescue *Festuca arundinacea*, buck's horn plantain *Plantago coronopus*, salt barley grass *Hordeum marinum* and boneseed *Chrysanthemoides monilifera* (Knox, 1992).

Many restorationists believe that to develop a wetland that will ultimately be a low-maintenance one, natural succession needs to proceed without human intervention - even if it means an initial period of invasion by undesirable weed species. Furthermore, if proper hydrologic conditions are imposed, such invasions will be temporary as water level fluctuations eliminate pest plant species in favour of desirable ones (Keddy, 1992; Mitsch and Gooselink, 1993). However, the correct hydrology and salt water flushing (in excess of that provided by the normal tidal regime) will only control species not normally associated with a tidal marsh i.e. *Festuca arundinacea*. Introduced salt marsh species (e.g. *Spartina alterniflora* and *Plantago coronopus*) will not be eliminated or prevented from establishing. They could thrive in this newly disturbed environment at the expense of native species which are not as vigorous or opportunistic. Such species will have to be weeded out mechanically and are more likely to be controlled at minimum levels rather than eliminated completely. By close spacing and pest plant control the rapid cover of initial native transplants can be enhanced. Herbicides, because of their lack of specificity and potential for on-going environmental damage, should only be used as a last resort and should be unnecessary if management is initially intensive.

Animal pest control

Rabbits are a significant threat to juvenile salt marsh plants especially those sourced from nursery stock (Chapter 4). Plants may be coated with repellent (80 gm egg powder, 150 ml acrylic resin and 800 ml warm water) (City Design, Christchurch City Council) when planted and consistent trapping should be carried out during the initial project stages until plants are sufficient in number and hardiness. Birds can be another threat to loosely rooted plants. "Mesurol" bird repellent may be necessary to prevent plant removal until securely rooted. Plants that have died or been removed may have to be replanted or replaced. Cats and dogs are an ever-present threat to the avifauna of wetlands and visitors should be discouraged from bringing them.

Litter and vandalism

These threats should be reduced if the public are sufficiently aware of, and involved in, project objectives. This can be achieved through school and community group involvement in monitoring and planting, news media and onsite signs. Fines also provide another deterrent. All litter should be removed. Sufficient lidded bins may also minimise this threat to both function and aesthetic appeal.

Nutrient deficiency

As the revegetation assessments show, fertilisation increases growth rate and plant success. However, given the elevated nutrient status of the Linwood Paddocks and the weed seed bank, fertilisation is probably unnecessary and undesirable, unless a specific nutrient is deficient.

7.12 Restoration Summary

Salt marsh restoration offers a positive option for the long-term use and environmental value of the Linwood Paddocks. Understanding salt marshes enough to be able to create and restore them requires a substantial training in plants, hydrology, soils, wildlife, water quality and engineering (Mitsch and Wilson, 1996). As each region has its own unique values, conditions and potential, a specific plan (locally researched) with defined goals is required. The primary goal and indicator of success, should be restoration of the functions of the ecosystem and not aesthetic appearance. If combined with the design for improved water quality in Chapter 6, the created marsh would have a gradient from fresh to salt water and should achieve all the restoration goals. The design is consistent with the Council's Green Edge proposal, and should enable restoration of a salt marsh wetland complex which is (i) representative of Canterbury salt marshes (ii) self-sustaining and (iii) accessible to the general public.

The restoration is mostly about revegetation and should be addressed in stages. It should be simple in design but not basis, and have an ecosystem approach. The design will be flexible and the marsh system will be given multiple opportunities through transplanting according to the revegetation template, and the establishment of a hydrologically open system, thus allowing nature to participate in the marsh design (Mitsch and Wilson, 1996). Once the site has been graded to the appropriate elevations (whilst still retaining Linwood Paddock topsoil) planting may begin. To increase salt marsh species' success, buffer species should be planted first, followed by salt marsh dominants that transplant well. If salt marsh herbs do not self-colonise, they may be planted once the initial rushes have established. Revegetation should be carried out in accordance with the revegetation template (Chapter 3). The species suggested for revegetation on the template are all native and occur naturally (either currently or historically) within the Canterbury region. All nursery stock must be grown from seed sourced locally, to ensure the correct ecotype for Canterbury and increase the likelihood of plant survival.

Consideration of the timing of marsh creation, marsh configuration, continuity with natural marshes, and attention to species habitat requirements should be sufficient to accelerate faunal colonisation of created marshes (Levin et al., 1996) and render deliberate faunal introductions unnecessary. Therefore, links with other systems (e.g. the Avon-Heathcote Estuary, the Oxidation Ponds, the Charlesworth Street Reserve and marsh areas of the Avon and Heathcote Rivers) are important and should be enhanced and maintained.

The development of the Paddocks in co-operation with the public will ensure that provision is made for the interpretation of natural values, and for public participation in management (Miller et al., 1994).

Continued monitoring is necessary to guide management and will focus mostly on maintaining the correct hydrological regime and controlling animal and plant pests.

8. GENERAL CONCLUSIONS

In these times of continued environmental degradation, scientific research is needed not simply for generating knowledge, but for decision-making for managers and environmental planners.

The aim of this study was to research Canterbury salt marshes in sufficient detail to provide a salt marsh ecological restoration and management plan specifically for the Linwood Paddocks. In addition to reestablishing functions and attributes normally associated with a salt marsh, the proximity to the Bromley Oxidation Ponds and the Avon-Heathcote Estuary provided the opportunity to investigate education, tourism and water purification functions, thereby creating a multi-purpose wetland complex.

Wetland design was determined not only by literature research and both quantitative and qualitative survey, but also by small-scale experimentation. The resulting restoration design is unique in New Zealand in that it has a detailed scientific basis. This research also contributes to the science of wetland restoration as a whole, ensuring that restoration becomes an applied ecological science and not a technique that is relearned each time without theoretical basis (Mitsch and Wilson, 1996).

Historical records of coastal Canterbury concerning vegetation composition and distribution were lacking in detail. Paintings and early reports merely reflected the early settler's perception of Christchurch as a flat area with large expanses of "swamp" that only had potential once drained, rather than an area with significant botanical diversity. Photographs and early maps proved more useful, since they indicated a dense and continuous salt marsh distribution in sheltered coastal areas.

Current salt marsh distribution and composition in Canterbury is similar to the patterns found worldwide; it is fragmented with respect to local distribution and elevational extent as a result of human impact. Surveys of the Avon, Heathcote, Brooklands and Saltwater Creek salt marshes enabled a continuous vegetation sequence to be pieced together for the coastal elevational gradient. This was complete and accurate enough to produce a revegetation template enabling efficient use of resources in restoration and a prediction of the likely vegetation pattern in a restored marsh (Chapter 3). The distinctive plant zonation observed here, as in salt marshes worldwide, appeared to be influenced by the interaction between the tide and elevation, which causes further variation in soil water salinity and the degree of anaerobiosis. The vegetation patterns observed in this study are similar to that recorded in salt marshes elsewhere in New Zealand e.g. Nelson (Davies, 1931) and Otago (Partridge and Wilson, 1988b). However, New Zealand has a particularly diverse coastline and there are important regional differences (Partridge and Wilson, 1988b) which need to be considered when applying this restoration design in other regions. For example, north of Tauranga the woody mangrove *Avicennia marina* forms the dominant coastal vegetation (Chapman, 1977) and the sea rush, *Juncus maritimus* (dominant in Canterbury) is virtually absent from Otago southwards (Partridge and Wilson, 1988b). In addition, each region has been affected differently along the elevation gradient by reclamation and pollution.

Salt marsh vegetation appeared to tolerate the observed pH, heavy metal and nutrient ranges recorded in (potentially) polluted marsh areas and Linwood Paddock soils. Further experimental tests of plant-tolerance to these factors would be required to determine the full tolerance range, rather than tolerance across the observed limited range of soil conditions. However, results were sufficient for this study where the aim was to determine the plant viability on Linwood soil.

The use of tidal mesocosms to determine species success in salt marsh restoration (Chapter 4) is a novel experiment in New Zealand. Although *Schoenoplectus* failed to regenerate following winter die-back, this was not entirely unexpected. Poor transplant success of this species has been observed in other studies (e.g. Partridge and Wilson, 1988b) and was probably caused by intolerance to transplanting rather than any negative effects of Linwood soil. Survival and growth of *Leptocarpus* and *Juncus* indicated that transplanting these species directly into Linwood topsoil would be a successful restoration strategy, especially if plants were pre-adapted to natural salt marsh conditions. If a further experimental site (also unlikely to be disturbed) but with a steeper and less confined elevation gradient were available, it would be useful to trial further species over a greater elevational (and, therefore, water, salinity and pH) range.

Investigation of freshwater species with the potential for phytoremediation of wastewater highlighted the need to consider local species and adapt overseas research to the New Zealand situation. Importation of species and expertise from outside New Zealand is only a temporary measure. The long-term solution involves immediate upgrading of the local capacity to utilise our own resources (Chua, 1997). Use of the exotic species *Phragmites* (currently trialed in Canterbury) poses a threat to biosecurity; use of native species (e.g. *Typha* or *Scirpus*) does not pose the same threat and should be used in preference. The exclusive use of native species in restoration will not only aid in the conservation of the species planted, but it will provide a more natural ecosystem and increase the habitat value for native wildlife. By using a combination of native species, conservative loading rates, high influent water quality and simple mechanical controls, such a wetland system could improve water quality and help meet final effluent limits as determined by the New Zealand Coastal Policy and the Regional Environment Plan. However, there are concerns about 'wetland' aging and decreased removal rates that will have to be addressed.

The attraction of New Zealand to the overseas tourist is the unique native wildlife, supposedly unspoilt nature, clean air, clean water and general lack of pollution. Therefore, preservation and promotion of our unique native wildlife has considerable commercial significance to the tourist industry. New Zealand has been slow to realise the value of enabling the tourist to view waterfowl from hides or the value of the diverse and interesting flora which wetlands support (Water and Soil Division, 1983). Salt marshes in particular provide the recreationist and the tourist with an open and changing vista, which adds to the variety and experience available within many localities throughout the country.

This restoration plan has been designed for persistence, recognising that it is necessarily a long-term landuse option. If each aspect of the restoration design, implementation and management is followed accurately, then the restoration should be successful in terms of reestablishment of salt marsh functions. The advantage with this design is that it relies on regular, minimal maintenance rather than capital and labour intensive inputs. Maintenance should be focussed on ensuring continuous hydrological links, since incorrect hydrology is one of the most serious threats to wetland success.

Assessments of previous local restoration attempts showed that post-construction maintenance is vital. Furthermore, monitoring and accurate record-keeping are vital to ensure that maintenance is purposeful and that parameters can be measured and techniques learned from such projects. On-going monitoring is also necessary to account for changing communities and to detect the effects of climate change and other environmental impacts.

Such a novel proposal - one that increases local botanical diversity, contributes to regional biodiversity, achieves protection from sea level rise and coastal erosion, serves to purify waste water and provides a unique tourist and educational complex – is surely an attractive and economic option for the use of Council-owned land currently under review. Indeed, it contributes towards achieving the discharge consents required of the Bromley Wastewater Treatment Plant, and is consistent with the new Christchurch City Council City Plan and Green Edge Proposal. Furthermore, successful restoration follows the principles of the Ramsar Convention, Agenda 21, the RMA and Department of Conservation policy.

Surely, if the misguided public and political perception of wetlands resulted in their loss and degradation, then an informed and educated society can reverse this trend and restore them. Indeed, as environmental issues begin to affect people's daily lives, increasing numbers of them are becoming concerned about the state of their natural environment and loss of biodiversity. Such concerns demand environmental accountability when redeveloping areas and making waste disposal decisions.

The guiding principles of such a scientifically sound proposal could be used within New Zealand and throughout the temperate world (with the application of local data) to ameliorate wetland loss. Restoration provides a range of functions greatly enhancing the environmental value of the area not only to the individuals who undertake the restoration, but also to society as a whole.

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